NITROGEN DYNAMICS OF AN ARABLE SOIL
UNDER
DIFFERENT AGRONOMIC PRACTICES

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PhD

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1991

I, Mark H. Redman, declare that this thesis was composed by myself.

Dr. A. Vinten planned and supervised installation of the hydrologically isolated plots at Boghall Farm and Glencorse Mains. Thereafter I conducted the majority of the field and laboratory work, receiving technical assistance with routine sampling and analysis from other members of the Soil Science Department, notably Carol Runciman and Rab Howard.

October 1991

ACKNOWLEDGEMENTS:

Sincere thanks are due to my supervisor, Dr. Andrew Vinten, for his sound guidance and gentle forbearance during the long gestation period of this thesis.

Thanks are also due to many other members of the Soil Science Department, past and present:

Carol Runciman and Ina Sutherland for diligent and patient technical assistance, Dr. Jonathon Arah for advice on the measurement of nitrous oxide emissions, Rab Howard for guidance and assistance in the use of the mass spectrometer for ^{15}N analysis, and Sara Wigglesworth for the soil incubation data she contributed from her BSc. Honours thesis.

The project was funded by the Department of Agriculture and Fisheries for Scotland.

ABSTRACT:

It has become evident that the benefits of increased fertiliser N use in the UK may be offset by problems, such as nitrate leaching to surface and groundwaters. The broad objectives of this work were to investigate how the N dynamics of a 'typical' arable soil in south-east Scotland receiving recommended fertiliser N applications were modified by: 1) reducing fertiliser N application; 2) replacing the fertiliser N with a leguminous source of N (forage peas grown as a green manure crop); 3) growing a winter cover crop.

All experimentation was field-based, with the main emphasis upon the direct measurement of NO_3-N leaching losses from eight 300 m^2 hydrologically isolated field plots, complemented by routine measurements of crop N uptake, soil mineral N, atmospheric N deposition and N_2O flux. N_2 fixation in the leguminous green manure was also measured, plus the mineralisation of the incorporated legume material. The efficacy of hydrological plot isolation in local soil types was first investigated using a small pilot plot.

The main experimental period began with incorporation of the green manure in September 1987 and ended in April 1989. Crop yields were low and the utilisation of applied N very poor. There was no apparent financial incentive to reduce fertiliser N application or replace it with a leguminous green manure.

Variable drainflow recovery from the plots hampered accurate estimation of NO_3-N leaching losses, but results suggested that: leaching losses from arable soils in south-east Scotland are generally less than in southern Britain; reducing fertiliser N application had little effect upon leaching losses; autumn incorporation of the green manure increased leaching during the following winter; autumn cultivation increased leaching compared with no cultivation; spring-applied fertiliser N was susceptible to leaching loss; growth of a winter cover crop may have reduced winter leaching. Denitrification was likely to have been a very important N loss process, but was very difficult to measure directly in the heavy, poorly structured soil type.

Despite very high levels of symbiotic N_2 fixation (over 300 kg N ha^{-1}), the use of the leguminous green manure to increase available N for the following crop was limited under local soil and climatic conditions. This was due to the complex nature of legume decomposition and mineralisation and led to the poor synchronisation of legume N release and crop N uptake in the autumn and spring after incorporation. It is likely, however, that a leguminous green manure would be of value in maintaining the long-term N status of an arable soil.

Experimental data was summarised in the form of N balance sheets for the different experimental treatments. These suggest that although the highest non-harvest losses occurred from the application of a recommended fertiliser N rate, this treatment retained mineral N within the arable soil-plant system most efficiently.

GLOSSARY OF ABBREVIATIONS:

See page number for full definition:

UNITS OF MEASUREMENT

Area:	m^2	square metre
	ha	hectare
Length:	m	metre
	cm	centimetre
	mm	millimetre
Mass:	t	tonne
	kg	kilogram
	g	gram
	mg	milligram
Time:	s	second
	hr	hour
Volume:	l	litre
	ml	millitre

OTHER

ARA	Acetylene Reduction Assay	
DM	Dry matter	
D_{corr}	Corrected weekly plot drainflow	124
CAP	Common Agricultural Policy	
CV	Co-efficient of variation	
EEC	European Economic Community	
L_{min}	Estimated mineralisation of legume material	103
L^*_{min}	Estimated mineralisation of legume material	103
LD_{corr}	Corrected weekly drainflow NO_3-N loading	124
LSR	Least Significant Range	104
MAFF	Ministry of Agriculture, Fisheries and Food	
N_{cr}	Nitrogen in residues from previous crop	182
N_{dfa}	Atmospherically-derived nitrogen	96
N_{dff}	Fertiliser-derived nitrogen	96,98
N_{dfs}	Soil-derived nitrogen	96,97
N_{dfl}	Legume-derived nitrogen	142
N_{hc}	Nitrogen in harvested crop fraction	
N_f	Fertiliser nitrogen input	
N_g	Nitrogen loss by denitrification	
N_k	Seed nitrogen input	
N_l	Nitrogen loss by leaching	
N_{min}	Estimated nitrogen mineralisation	101,182
N^*_{min}	Estimated nitrogen mineralisation	102
N_{om}	Soil organic nitrogen content	
N_p	Rainfall nitrogen input	
N_{rt}	Nitrogen remaining in crop roots	182
N_s	Soil mineral nitrogen content	
NH_4-N	Ammonium-nitrogen	
NO_3-N	Nitrate-nitrogen	
s.e.	Standard error	
TN_{min}	Estimated total nitrogen mineralisation	101
TN^*_{min}	Estimated total nitrogen mineralisation	102
WHO	World Health Organisation	

CONTENTS:

CHAPTER 1 : INTRODUCTION

Sustained agricultural production depends upon the continual fixation of atmospheric nitrogen (N_2) to replenish the N lost in harvested crops, livestock production and natural processes such as nitrate leaching and denitrification.

Throughout much of agricultural history this N_2 fixation was achieved biologically by the inclusion of legumes in crop rotation. In a review of ancient agricultural practices, Semple (1928) indicated that several early writers specifically discussed the use of legumes for soil improvement. For example, the Roman statesman Cato (234-149 B.C.) acknowledged the value of leguminous plants in raising soil fertility, both directly and through the provision of manure derived from their use as livestock feed.

In Britain crop rotations including leguminous crops formed the traditional base of agriculture for many centuries. One of the best known rotations in the 1700s was the Norfolk four course. This originally took the form of:

Roots - Barley - Seed - Wheat

where the seed part of the rotation was some form of legume, notably a one-year red clover ley (sometimes with ryegrass) or an arable legume crop. In some parts of the country the one-year seeds crop was extended into a short-term two-year ley, or into the medium- to long-term leys which form the basis of traditional 'ley/arable' farming systems *e.g.* 4-5 year forage legume or grass/clover ley followed by up to 3 years cereals (Laity 1947).

There was no extensive investigation of the contribution of legumes to soil fertility until the mid-1800's (Russell 1966), but by the 1920's processes such as N_2 fixation and organic N mineralisation were reasonably well understood (*e.g.* Löhnis 1926, Pieters 1927).

Between the late 1940s and the early 1980s UK agriculture underwent rapid modernisation as government and EC support created a favourable economic climate in which technical efficiency and technological advancement were encouraged. With the increasing availability of cheap supplies of industrially fixed fertiliser N the necessity for, and emphasis upon, legumes in crop rotations inevitably declined.

Between 1950 and 1982 fertiliser N use by UK farmers increased by 700-800% contributing to a steady rise in agricultural productivity (Jollans 1985). Hood (1982) suggested that fertiliser N was responsible for 30-50% of crop yield increases, assisted by other factors such as improved varieties, increased agrochemical inputs and better husbandry techniques.

In 1985-86 the total annual input of N into UK agriculture was estimated as 2.4 million tonnes, of which only 0.3 million tonnes came from biological N_2 fixation, whilst 1.6 million tonnes came from fertilisers. In addition, at least 0.3 million tonnes came from atmospheric deposition and 0.2 million tonnes N was imported as animal feedingstuffs and human food (Jenkinson 1986). During this period the UK was the second largest user of fertiliser N within the EEC, with the third highest average N use per hectare (132 kg N ha^{-1}) (FAO 1986). It was concluded by the Royal Society (1983) that not only would fertiliser remain the principle source of N for UK agriculture into the foreseeable future, but that the annual rate of application was likely to continue rising.

Increases in fertiliser N use have not occurred uniformly across the UK, but have tended to be concentrated in those areas of the country most suited to intensive agricultural production. These areas have seen significant changes in agricultural land use, notably a decline in traditional ley/arable rotational systems and an increase in continuous arable cropping. In much of eastern England agricultural land use is now dominated by intensive cereal production, interrupted only by occasional break crops (*e.g.* Table 1.1).

Table 1.1 : Percentage distribution of agricultural land use in East Anglia in 1986 (only crops accounting for more than 5% of total land use are shown)

Cereals	Grass < 5 yr^{-1}	Grass > 5 yr^{-1}	Arable legumes	Sugar beet	Horticulture
56.0	2.0	8.5	3.0	11.0	5.0

Source : 1986 MAFF census data

In 1985 over 40% of the total fertiliser N input to UK agriculture was applied to arable crops in England and Wales, with average annual application rates ranging from 102 kg N ha^{-1} for spring barley to 272 kg N ha^{-1} for oilseed rape (Jenkinson 1986). Equivalent data for Scotland is not readily available, but it is interesting to note that in much of Scotland ley-based crop rotations still remain an important part of many farming systems (*e.g.* Table 1.2).

According to Jollans (1985), farmers use N fertilisers to improve the profitability of their farming operations, whilst consumers benefit from the lower food prices which fertiliser use makes possible, and the nation as a whole benefits from the security of increased food supplies. In 1979 the official view was that "a sustained increase in agricultural net product is in the national interest and can be achieved without undue impact on the environment" (Farming and the Nation, White Paper, cited in Jenkins 1990). Today, however, it is

Table 1.2 : The occurrence of temporary (*i.e.* rotational) grassland on farms in south-east Scotland

Grassland in rotation	Proportion of cropping farms (%)
no grass	10
1 year grass	15
2-3 year grass	70
4-5 year grass	5

Source: ESCA (1982)

evident that the increases in agricultural production achieved may be offset by some problems. In particular, the increased extent and intensification of arable cropping in the UK is likely to have contributed to a greater 'leakage' of N from agricultural land into other parts of the environment where it is not so welcome.

Before moving on to discuss this issue in more detail, it is necessary to briefly consider how N moves through the arable soil-plant system.

1.1 THE N DYNAMICS OF AN ARABLE SOIL

The N dynamics of an arable soil-plant system can be defined as 'the flow of N through, and within, the crops and soil of a predominantly arable cropping regime'. In some cases (*i.e.* all-arable farms) this will encompass the whole farm N cycle; in many other cases the N dynamics of the arable system should be viewed as a discrete sub-component of a larger and more complex farm N cycle.

Figure 1.1 outlines the potential N dynamics of an arable soil, broadly classifying them as external inputs, transformations and losses.

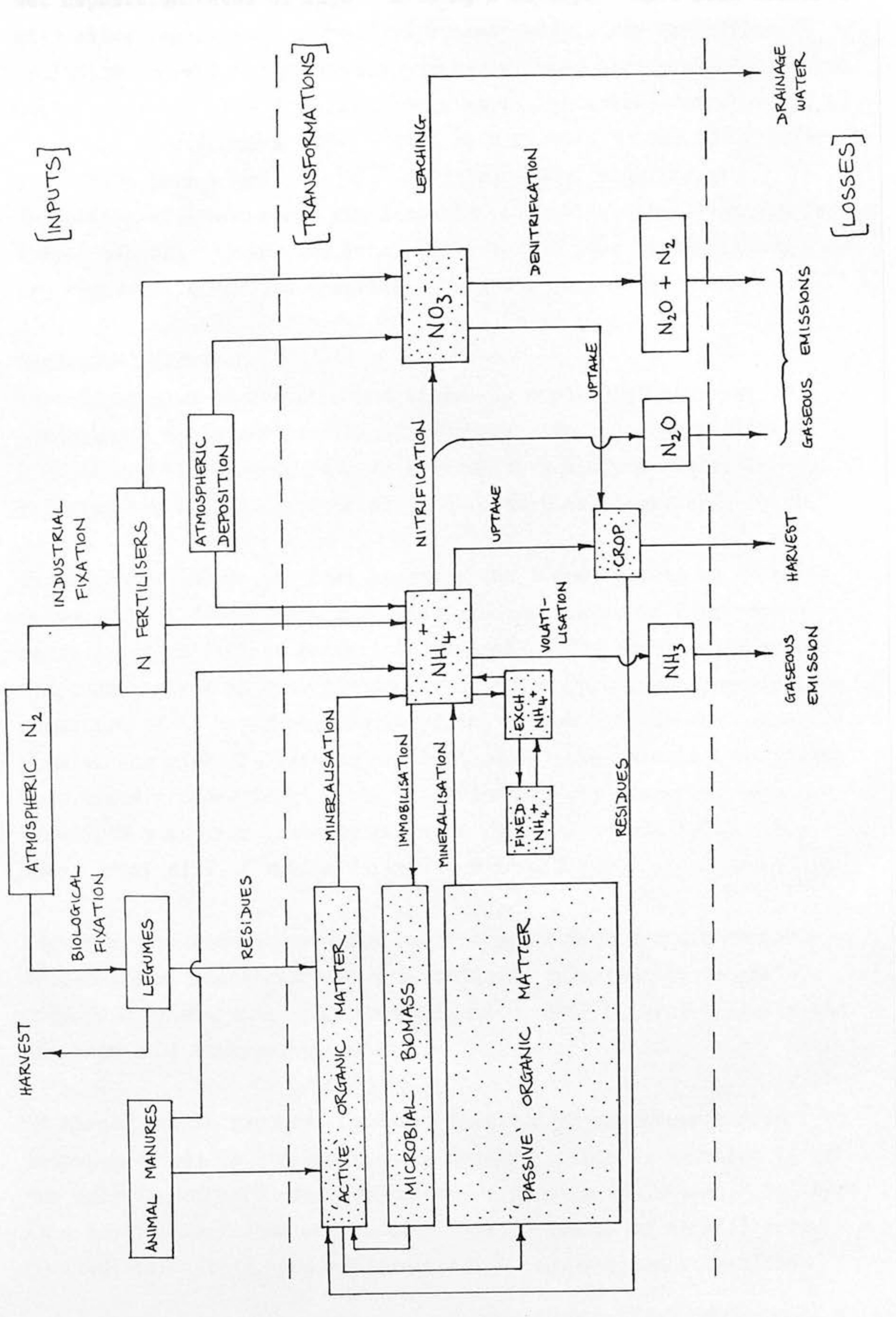
1.1.1 EXTERNAL N INPUTS

N can enter the arable soil-plant system in a number of different ways: atmospheric deposition, biological N_2 fixation and fertilisers effectively import N from outside of the farm, whilst animal manures transfer N from other parts of the farm's N cycle (*e.g.* livestock enterprises on grassland).

Atmospheric deposition

External N inputs can occur via the deposition of N compounds (NO, NO_2, HNO_2, HNO_3, NH_3) from the atmosphere. The major source of N oxides is fuel combustion and the so-called NO_x emissions from power stations and motor vehicles (Royal Society 1983). Relatively small amounts of N are also fixed by lightning. Atmospheric NH_3 is derived from a number of sources including industrial emissions, coal burning, agricultural soils (1.1.3) and manures (Bouwman 1990b).

Figure 1.1: The potential N dynamics of an arable soil



Since most atmospheric N compounds are highly soluble in water, significant deposition of NO_3 and NH_4 often occurs in rainfall *e.g.* wet deposition rates of 11.6 - 17.5 kg N ha^{-1} yr^{-1} have been measured at 3 sites in southern England (Williams 1976). Dry deposition, including gaseous and particulate material, may also occur (Söderlund 1981). This is very difficult to measure, but annual rates of 4-5 kg N ha^{-1} as NH_3 (Rodgers 1978) and 12 kg N ha^{-1} as NO and NO_2 (Fowler 1978) have been suggested. In localised areas, significant deposition of atmospheric NH_3 can occur following volatilisation from animal manures (Speirs and Frost 1987) and in some circumstances this may represent a novel N transfer within the farm N cycle.

Biological fixation

Certain species of bacteria and algae are capable of reducing atmospheric N_2 to ammonia (NH_3). The most important agricultural example are the *Rhizobia* bacteria which form a close symbiotic relationship with leguminous crops such as peas, beans and clover.

The bacteria invade the root hairs of the legume plants at an early stage of root development and stimulate the plants to form root nodules (which enclose bacterial colonies). The host plants supply the nodules with photosynthetic carbon (C) to provide energy for the reduction of N_2 to NH_3 by the bacterial enzyme nitrogenase. The nodules transfer NH_3 back to the host plant (via amination and amino acid transfer reactions) where it is immediately converted into organic N compounds, notably proteins (Havelka *et al.* 1982). The measurement of N_2 fixation is reviewed in 2.2.

Legume crops often leave large quantities of high protein content crop residues which can make a significant contribution to soil organic N levels and, upon mineralisation, soil mineral N levels and the growth of subsequent crops.

It should not be assumed, however, that all of the organic N in legume residues is atmospherically-derived, since N_2 fixation is not the only source of N used for protein synthesis in legumes. If there is a readily available source of mineral N (soil- or fertiliser-derived) then it is more efficient for a legume plant to utilise this, rather than support N_2 fixation *i.e.* the energy cost of bacterial N_2 fixation is about 250 GJ t^{-1} N fixed, compared to 215 GJ t^{-1} soil mineral N taken up by plant roots and metabolised (King 1983). In reality this means that, unless grown in conditions of very low soil N status, legumes in an arable system may not fix as much N_2 as might be expected. The agronomic value of legumes is is discussed further in 1.4.

Fertilisers

The industrial fixation of atmospheric N_2 is analogous to biological fixation since both processes require energy to reduce N_2 to NH_3. During the commercial manufacture of NH_3 (Haber-Bosch process), hydrogen and atmospheric N_2 are combined at high temperature (300-500°C) and pressure (400-1000 atmospheres) in the presence of a catalyst such as reduced iron (Haynes 1986). The industrial production of NH_3 is extremely efficient at about 36 GJ t^{-1} N, compared to the ultimate thermodynamic limit of 28 GJ t^{-1} N (King 1983).

The NH_3 produced in the Haber-Bosch process can be used either directly as fertiliser N, or more commonly after processing to ammonium nitrate or urea ('straight' fertilisers) or mono- and di-ammonium phosphates (used in 'compound' fertilisers) (Brockman 1982). Compared to the use of legumes as an N source, fertilisers *directly* supplement soil mineral N levels and therefore rapidly increase the amount of N available for crop uptake.

Animal manures

Of the N consumed by livestock in the form of herbage and concentrate feeds, a relatively small proportion is actually utilised for the production of meat or milk. The majority of the N is excreted as dung and urine, of which, in the case of cattle, about half is collected in yards and buildings as slurry and manure (Whitehead *et al.* 1986). Some of this will be subsequently applied to arable crops.

Manure N comprises two major fractions of agronomic interest, ammoniacal and organic N (Beauchamp and Paul 1989). Ammoniacal N is water soluble, comprising of urea and ammonium, and supplements soil mineral N levels in a similar manner to fertilizer N. The organic N fraction must undergo mineralisation before being available for crop uptake.

1.1.2 NITROGEN TRANSFORMATIONS

Soils contain N in two discrete pools: organic and inorganic. Depending upon the soil and environmental conditions, agricultural soils contain 1 000 - 6 000 kg N ha^{-1} (White 1979), almost all of which is in organic form (C:N ratio approximately 10:1) and unavailable for crop uptake.

Soil organic matter is colonised by a variety of heterotrophic soil organisms which derive their energy for growth from the decomposition of organic molecules. During decomposition essential nutrient elements are also converted from organic to inorganic forms. This is termed mineralisation and occurs whenever soils are moist and warm enough for microbiological activity, with a 'flush' of intense activity usually occurring in the spring and to a lesser extent in the autumn (Cooke 1976).

Soil organic matter can be broadly classified into two types (Jansson and Persson 1982):

a) **active phase** - this comprises the fresh plant, animal and microbial debris which is most easily mineralised. The commonest organic materials entering an arable soil are, with a few exceptions, crop residues. These provide the *primary input* of organic matter, whilst the remains of dead soil organisms form the *secondary input* (Jenkinson 1981).

b) **passive phase** - this comprises those constituents of the active phase which show some resistance to mineralisation. These residues accumulate, often undergoing biological or chemical modification, to become the humus fraction of the soil. Mineralisation of this fraction is relatively slow, but because it is present in such large quantities it is still a significant source of crop nutrients.

The extent of the passive organic phase (humus content) is determined by an equilibrium between the accumulation and decomposition of organic residues. This equilibrium is influenced by many factors including soil texture, climate, cropping system and residue management. For example, in the cool Scottish climate the equilibrium tends towards accumulation and local soils are characterised by higher organic matter levels than occur in more southerly parts of Britain.

The mineralisation of organic N involves the degradation of proteins, amino acids, nucleic acids and other nitrogenous compounds to the ionic species NH_4^+. Once formed, NH_4^+ joins the inorganic or mineral pool (along with NH_4^+ from fertiliser, manure and atmospheric deposition) and has a number of possible fates (Paul and Clark 1988):

1. In the presence of readily available carbonaceous material, a proportion of the NH_4^+ is rapidly assimilated into newly formed microbial biomass *i.e.* N immobilisation inevitably accompanies N mineralisation. Subsequently the microbial biomass dies, enters the active soil organic matter phase and becomes liable to decomposition.

   The continuous transfer of mineralised N into microbial biomass, and of immobilised N back into inorganic N has been defined as MIT (mineralisation-immobilisation turnover) and forms a heterotrophic sub-cycle within the overall dynamics of the soil-plant system (Jansson and Persson 1982). The result of MIT is a net effect - net mineralisation or net immobilisation - which influences the N supply for other N cycle processes;

2. When (as in usual soil circumstances) microbial development is limited by available carbon (C), most of the mineralised NH_4^+ is oxidised as rapidly as it is formed to NO_3^- by the process of nitrification (Schmidt 1982).

   This is a two-stage process mediated by 2 groups of chemoautotrophic bacteria. The first group includes the genus *Nitrosomonas* and oxidises NH_4^+ to NO_2^-, whilst the second group includes the genus *Nitrobacter* and oxidises NO_2^- to NO_3^-. Oxidation of NO_2^- is more rapid than that of NH_4^+, so there are only ever trace amounts of NO_2^- in the soil;

3. NH_4^+ can be adsorbed onto the surface of clay minerals and soil organic matter, from where it is freely exchangeable with other cations in the soil solution;

4. NH_4^+ is approximately the same size as the K+ ion and readily enters the interlayer portions of clay minerals. The collapse of the interlayer space, for example by drying, effectively fixes the NH_4^+ making it only very slowly available to exchange with the soil solution;

5. NH_4^+ can be taken up by plants, although the chemical and biological processes already described generally make NO_3^- the most prevalent form of soil mineral N in well-drained soils. Consequently, the majority of crops have evolved mechanisms for absorbing NO_3^- (Olson and Kurtz 1982) and if crops do absorb NH_4^+ this tends to be early in the growing season when NH_4^+ predominates because nitrification is limited by low temperatures;

6. Under certain circumstances NH_4^+ can be lost from applied N fertilisers by volatilisation as gaseous NH_3 (1.1.3).

NH_4^+ occupies the soil mineral N pool along with the NO_3^- ion. Much of the NO_3^- will be derived from the nitrification of NH_4^+ (see above), but levels are also supplemented directly by significant quantities of fertiliser NO_3-N and, to a lesser extent, atmospheric deposition.

As already noted, NO_3^- is the most important source of N for crop uptake, but it is also susceptible to significant losses from the soil (1.1.3) and this makes it of great environmental, as well as agronomic importance.

1.1.3 N LOSSES

In addition to N losses via crop uptake and removal at harvest, soils can lose N in four specific ways:

1. Nitrate Leaching

Arable soils can 'leak' substantial quantities of NO_3^- in drainage water and this is frequently cited as the most important channel of N loss, other than that accounted for in crop uptake (Legg and Meisinger 1982). Major losses occur when two conditions are met:

(a) *Soil water movement is large i.e. the influx of water (either rainfall or irrigation) is greater than the evapo-transpiration.*

This is further influenced by soil texture and structure which affect the hydraulic conductivity and water storage capacity of a soil. NO_3-N leaching losses are generally greater from poorly structured sandy soils than heavily structured clay soil (Haynes 1986);

(b) *Soil* NO_3^- *levels are high due to the mineralisation of organic N, or the presence of excessive or unused fertiliser.*

The proportion of leached NO_3^- derived from organic or fertiliser N sources will depend upon environmental conditions, the rate of applied N and the crop management systems employed. However, considerable quantities of NO_3^- leached from arable soils can originate from soil organic N rather than directly from applied fertiliser (*e.g.* Dowdell *et al.* 1984).

Seasonal rainfall and evapo-transpiration patterns interact with soil NO_3^- levels to affect leaching losses. Although these vary greatly from year to year, and between regions, some general statements can be made (*e.g.* Haynes 1986, Powlson 1988):

a) in summer evapo-transpiration generally exceeds rainfall and leaching is usually minimal. However, fertiliser N losses can occur if application coincides with intense, heavy rainfall;

b) a particularly dry summer can limit crop N uptake and lead to the accumulation of NO_3^- in the soil which is then susceptible to autumn/winter leaching;

c) provided that the soil is approaching or has returned to field capacity, autumn rainfall will leach any NO_3^- remaining from pre-harvest fertiliser application or derived from late summer/autumn mineralisation. Applications of fertiliser N to the seedbed of autumn sown crops are also very susceptible to leaching;

d) in winter there is a large excess of rainfall over evapo-transpiration and any NO_3^- present in the soil profile is readily leached;

e) spring-applied fertiliser N is susceptible to leaching if application coincides with heavy rainfall or NO_3^- is not rapidly removed by crop uptake.

2. Biological Denitrification

Denitrification is the dissimilatory reduction of NO_3^- to the gaseous forms of nitric oxide, NO, nitrous oxide, N_2O, and molecular nitrogen, N_2. It is the major natural process by which oxidised N is returned to the atmosphere (Royal Society 1983).

Denitrification occurs under anaerobic soil conditions, when NO_3^- replaces O_2 as the terminal electron acceptor in microbial respiration. Conditions favouring denitrification are the presence of: adequate NO_3-N levels, denitrifying organisms, high soil water contents and poor soil structure (both of which result in low air filled porosity and potentially anaerobic conditions). The main rate determining factors are soil temperature and the amount of readily available carbon substrate present (see reviews by Firestone 1982 and Knowles 1981).

Denitrification can be a very important N loss process in arable soils, particularly on heavy soils, but its measurement is frequently complicated by high spatial and temporal variability (Folorunso and Rolston 1984) and the difficulties of determining total denitrification loss from the measurement of N_2O emissions (Arah *et al.* 1991).

3. NH_3 **Volatilisation**

Gaseous losses of NH_3 are much more serious from agricultural systems involving livestock than from arable agriculture (Whitehead *et al.* 1986). In arable soils, the most significant losses occur when ammoniacal fertilisers or urea are applied under alkaline conditions, for example to the surface of calcareous soils.

NH_3 losses resulting from surface volatilisation are aggravated by high soil temperatures and drying conditions, but can largely be prevented by placing fertilisers below the soil surface or working them in thoroughly with the top soil (Tisdale and Nelson 1975).

4. N_2O **Emission during Nitrification**

The importance of nitrification as a source of N_2O emissions from soil has only been fully appreciated in the last decade or so. Evidence for N_2O loss during nitrification comes from the observation that some soils evolve N_2O even when they are well aerated and their moisture content is low (*i.e.* under conditions known to inhibit denitrification), and that N_2O emissions in these circumstances are correlated with nitrifiable-N contents rather than NO_3-N levels (see review in Bouwman 1990b).

1.2 NITROGEN AND THE ENVIRONMENT

N has been described as the "black sheep of the nutrient family" since, although essential to plant growth and successful crop production, it also has the potential to cause ecological disturbance and possibly risk human health when 'leaking' from agricultural systems into neighbouring natural and semi-natural environments.

Greatest concern is over the pollution of water resources with nitrate-nitrogen (NO_3-N) draining from agricultural land and this is discussed in most detail here. Gaseous N losses from agriculture are also discussed, but their full impact upon the environment still remains unclear.

1.2.1 NITRATE POLLUTION OF WATER RESOURCES

During the last 20 years levels of NO_3-N in many UK ground and surface waters have been gradually rising (HMSO 1986).

An analysis of 12 rivers for which data was available over a 20 year period showed increases in NO_3-N concentration of between 50 and 400% (Wilkinson and Greene 1982). Rivers with the highest NO_3-N levels are found in the Midlands and south-east England, with the lowest levels in the mountainous regions of Wales, northern England and Scotland. There is, for example, apparently no NO_3-N pollution problem in the River Forth catchment area, which is the major river system in central and south-eastern Scotland (F.R.P.B. 1989). The temporal and regional trends in river water quality are also reflected in lakes and reservoirs (Royal Society 1983). For example, there has been a noticeable increase in reservoir NO_3-N levels in south-east England.

Considerable quantities of freshwater are stored in groundwater aquifers, the most important of which are the chalk and Triassic sandstones. Available long-term data on the NO_3-N concentration of groundwater are less common than for surface waters. Nevertheless there is a marked upward trend in many catchment areas, particularly in the dry eastern areas of England (Wilkinson and Greene 1982, Foster 1976).

There are two major concerns about the nitrate pollution of water resources: eutrophication in surface waters and the quality of drinking water.

Eutrophication

Many surface waters, such as rivers and lakes, have a limited supply of N and phosphorus (P) and are described as oligotrophic (nutrient poor, low biological productivity). Enrichment of these waters with an available source of N and/or P, notably from agriculture or domestic sewage, can transform them into being eutrophic (nutrient rich, high biological activity). For example, in Scotland several freshwater lochs in the urbanised midland belt and productive agricultural areas of the south-east have become eutrophic (*e.g.* Loch Leven and Forfar Loch), whereas those in the north and west of the country have generally remained oligotrophic (Stewart *et al.* 1976).

Problems associated with eutrophic waters are (Cooke 1976):

a) surface algal blooms which detract from the appearance of waters and impair their amenity value. Blooms are also often responsible for taints in public water supply and problems with filtration, consequently increasing the cost of water purification when drawn from surface waters. Decomposing algae can remove so much oxygen from the water that fish die;
b) increased growth of rooted water weeds can cause difficulties with river management, flood control and navigation, whilst decaying weeds can cause taints and the death of fish.

In most freshwaters N is more abundant than P and the main cause of eutrophication is generally considered to be increased P levels (*e.g.* Jollans 1985, Royal Society 1983). However, there are conflicting opinions on this issue, notably from the NCC (HCEC 1987) and the Standing Technical Advisory Committee on Water Quality (STAWQ 1984), whilst the Nitrate Co-ordination Group (DoE 1986) considered that in some cases small increases in nitrate have been shown to produce marked changes in the diversity and productivity of algae in lakes. NO_3-N is, for example, considered to be the specific cause of eutrophication in Loch Leven in Kinross-shire.

Concern is also being expressed over the increasing incidence of marine eutrophication in the coastal waters of the North Sea and it is thought that this is linked to N rather than P levels (Barrett 1988). Again the exact processes involved remain unclear, with the Nitrate Co-ordination Group (DoE 1986) suggesting that NO_3-N enhances the extent of marine algal blooms rather than actually initiating algal growth.

Nitrate and drinking water quality

Ingestion of large amounts of nitrate in drinking water may be harmful to humans. Nitrate itself is relatively non-toxic and is rapidly excreted from the body (Magee 1982). However, nitrate may be partially reduced to nitrite by bacteria in the mouth and gut, and it is this nitrite that is potentially toxic.

The main alleged health hazards of nitrate in drinking water are methaemoglobinaemia in young babies and gastric cancer.

1. **Methaemoglobinaemia**

   It has been known since the 1940s that excessive quantities of nitrate in drinking water present a health risk to young (under 3 months) artificially-fed babies (Fraser and Chilvers 1981, Owen and Jürgen-Gschwind 1986). Nitrite, derived from the bacterial reduction of ingested nitrate, is absorbed into the bloodstream where it combines with haemoglobin to form methaemoglobin which cannot transport oxygen. This condition becomes manifest as clinical cyanosis ('blue baby syndrome') and can lead to death.

   In the UK only 10 cases (including 1 death) have been reported since the illness was first described (Owen and Jürgen-Gschwind 1986). Globally, some 2 000 cases have been reported (with a case fatality of about 8%), many of which were associated with bacterially contaminated private water supplies which accentuated the disorder (Fraser and Chilvers 1981).

2. **Gastric cancer**

   Under certain conditions, nitrites react with various amines to form nitrosamines and other N-nitroso compounds. Most of these compounds are strongly carcinogenic in animals, although none have yet been directly implicated as the cause of human cancer (Magee 1982). The site commonly regarded as being at risk from nitrosamines is the stomach. Stomach acidity favours nitrosation reactions and many nitrosamines are known to be locally acting in animal experiments (Forman *et al.* 1985). It has therefore been suggested that an increase in human nitrate intake, both from drinking water and foods such as vegetables, may lead to an increased risk of gastric cancer.

   The evidence available from international epidemiological studies on nitrate ingestion and gastric cancer is, however, inconclusive (references cited in and including : Fraser and Chilvers 1981, Forman *et al.* 1985). Furthermore, the actual contribution of water-borne nitrate to human nitrate intake remains debatable. It has been established, for example, that unless the nitrate concentration of drinking water is especially high, food can represent a more significant source of dietary nitrate than water (RCEP 1979).

The current World Health Organisation (WHO) European Standards for Drinking Water recommend levels of less than 11.3 mg NO_3-N l^{-1}, with 11.3-22.6 mg NO_3-N l^{-1} regarded as acceptable, and levels over 22.6 mg l^{-1} as not recommended. In 1980 the EEC adopted a community directive specifying stringent limits for NO_3-N in drinking water supplies, with a guide value of 5.7 mg NO_3-N l^{-1} or below and a 'maximum allowable concentration' of 11.3 mg NO_3-N l^{-1} (Wilkinson and Greene 1982).

It has been estimated that about 3.2% of the UK population receives water exceeding 11.3 mg NO_3-N l^{-1} at any time (Owen and Jürgen-Gschwind 1986). Some of the rivers in eastern England that are used for potable water supply have, for short periods, exceeded the W.H.O. maximum recommended limit (22.6 mg NO_3-N l^{-1}), and supply abstraction has been curtailed. At least 100 groundwater sources in the UK have either continuously or intermittently exceeded the W.H.O. acceptable limit (11.3 mg NO_3-N l-1) and the most severely affected wells have been withdrawn from service (Wilkinson and Greene 1982). For example, levels of 23 and 27 mg NO_3-N l^{-1} have been reported for chalk and Triassic sandstone aquifers respectively (Central Water Planning Unit 1977). About 16% of private farm water supplies surveyed by Webber and Wadsworth (1976) exceeded 11.3 mg NO_3-N l^{-1}.

Most Scottish groundwaters are potable, although rising NO_3-N levels have been identified in some aquifers, whilst some wells and boreholes have exceeded the EEC 'maximum allowable concentration' notably in eastern and south-eastern Scotland (Robins 1986). Groundwater sources only provide about 2% of Scottish water consumption and NO_3-N pollution is not considered a major problem. However, where groundwater is the sole water supply, for example in private supplies to isolated farms and communities, it has been recommended that NO_3-N levels in drinking water should be regularly checked (McLarty 1980).

Agriculture and nitrate pollution of water resources

There is little doubt that the increased levels of NO_3-N in freshwater resources have resulted mainly from the drainage of agricultural land (Royal Society 1983). This has been linked with the increased use of N fertilisers (Foster *et al.* 1982, Smith 1976), as well as changes in land use, particularly the ploughing of grassland (Young and Gray 1978, Cameron and Wild 1984). The exact nature of the relationship between agriculture and water resource quality is, however, variable.

In the first instance, the relationship between agriculture, and ground and surface water is quite different. The retention time of water and solutes in groundwater is very high, often exceeding 10 or 20 years. Therefore the effects of pollution incidents in groundwater are effectively buffered, becoming apparent more slowly and persisting much longer than in surface waters (Foster *et al.* 1982).

The vertical profiles of NO_3-N in the unsaturated zone above aquifers are a useful means of evaluating the recent history of leaching losses from overlying agricultural land (*e.g.* Foster *et al.* 1982, Young and Gray 1978). Profiles from unfertilised and fertilised grassland are characterised by low uniform NO_3-N concentrations, frequently less than 2 mg NO_3-N l^{-1}. Profiles under long standing arable soils are characterised by a major 'nitrate front', with concentrations well above W.H.O. limits, at around 3-8 m below ground level.

In 1963 and 1964 thermo-nuclear fall-out led to significantly higher tritium levels in rainfall. It has thus been possible to follow the movement of water through the profile of the unsaturated zone by measuring the tritium peak. Available data suggests an infiltration rate of between 0.3 and 1.0 m year^{-1} in the chalk and 2.0 m year^{-1} in the Triassic sandstone (Royal Society 1983). If it is assumed that NO_3-N and tritium move at the same rate down the profile, then it would appear from the relative positions of the NO_3-N and tritium peaks, that there has been a steady increase in NO_3-N leaching losses from arable land into aquifers since the 1960's (Royal Society 1983).

It can not be assumed, however, that fertiliser N use is solely responsible for increased aquifer NO_3-N levels, since whilst fertiliser N application was increasing there was also an increase in the amount of grassland converted to arable land, particularly in the arable areas of eastern England (Foster *et al.* 1982). The mineralisation of ploughed in grassland is potentially a very important source of NO_3-N for leaching (*e.g.* Low *et al.* 1963), and can have a marked effect upon NO_3-N profiles, leading to peaks in concentration of up to 50 mg l^{-1} (Cameron and Wild 1984, Foster *et al.* 1982). It is therefore difficult to distinguish with certainty the relative contribution of fertilisers and organic matter mineralisation in increasing groundwater NO_3-N concentrations.

Surface waters show considerable year-to-year and seasonal variations in NO_3-N concentration. For example, NO_3-N concentrations are often lowest during the summer months and highest during the winter (Casey and Clarke 1979, Stewart *et al.* 1976). Owens (1970) identified land drainage as the major source of nutrients in rivers, with more NO_3-N contained in drainage from arable land than grassland, whilst good correlations have been reported between mean river NO_3-N concentration and fertiliser N use (Smith 1976). Streams are very susceptible to local agricultural practice, particularly if heavy rain immediately follows the application of fertiliser or organic manures (Webber and Wadsworth 1976). This susceptibility to pollution can be accentuated by local topography and soil type, leading for example to rapid downslope leaching losses (Troake *et al.* 1976).

In a study of Loch Leven, a eutrophic surface water in eastern Scotland, Holden (1976) suggested that 85% of the N input into the loch was derived from agriculture. Increased NO_3-N levels in the tributary streams of the Loch between 1964 and 1973 corresponded to a 50% increase in fertiliser N use over the same period. In a further study of Loch Leven by Cuttle (1982), arable land was identified as the main source of N in the loch with significant N inputs, not only from fertiliser, but also livestock wastes and ploughed grassland.

Whilst the broad trends linking agriculture and water quality have been described here, comparative leaching losses from different farming systems and management regimes can only really be confidently investigated on a field scale.

1.2.2 ATMOSPHERIC POLLUTION BY GASEOUS N COMPOUNDS

N_2O and NH_3 emissions from soil are potential sources of atmospheric pollution and have both been implicated as a threat to the environment.

Although N_2O is chemically inert in the troposphere, it is capable of absorbing infra-red radiation and is therefore an effective 'greenhouse' gas. The contribution of N_2O to the supposed temperature rise associated with increased global warming over the past 100 years is claimed to be about 4%, with atmospheric levels currently increasing at about 0.25% per year (Bouwman 1990a).

Very little N_2O returns to the earth's surface. Most is photochemically destroyed in the stratosphere where it is an important source of nitric oxide (NO) which has a strong catalytic effect upon ozone depletion (Keeney 1982). The stratospheric ozone layer plays an essential role in reducing terrestrial exposure to ultraviolet radiation by filtering out short wavelength radiation from the solar spectrum. An increase in solar ultra-violet radiation at the earth's surface could have several effects, including an increase in the incidence of human skin cancer (Magee 1976) and unpredictable eco-system disturbance (Keeney 1982).

There is thus a possible link between increased use of fertiliser N, increased N_2O emissions from denitrification activity, and increased global warming and ozone depletion (Wellburn 1988, Bouwman 1990b).

NH_3 can absorb infra-red radiation, but its role in global warming is not significant due to its short atmospheric residence time (Bouwman 1990b). Most volatilised NH_3 returns to the earth's surface where it represents a N input to agricultural systems (1.1.1), but is also of increasing concern to environmentalists since it has been identified as a possible factor in the extinction of N-sensitive plant species in low fertility ecosystems (Ellenburg 1987). Furthermore, NH_3 deposition plays a major role in the process of soil acidification (*e.g.* Speirs and Frost 1987) which is undesirable in many natural and semi-natural ecosystems.

1.3 POTENTIAL CHANGES IN AGRICULTURAL N PRACTICE

Wagstaff (1987) suggested that changes in agricultural N practice are only likely to arise due to imposed restrictions and accepted codes of practice, or the adoption of a fundamentally different approach to farming. These factors are already having an effect upon farmers' N use in the UK.

1.3.1 IMPOSED/VOLUNTARY RESTRICTIONS

To help deal with the problem of nitrate pollution, the Government introduced a pilot scheme (under the 1989 Water Act) in July 1990 to promote the adoption of agricultural methods designed to reduce nitrate leaching (MAFF 1990a).

Under the scheme 10 Nitrate Sensitive Areas (NSAs) were designated where farmers were offered 'Basic' payments to reduce nitrate leaching within existing agricultural practices (*e.g.* limit organic and inorganic nitrogen applications, and refrain from ploughing permanent grassland) and 'Premium' payments to make more fundamental changes (*e.g.* conversion of arable land to unfertilised grassland) (MAFF 1990a). Participation in the NSA scheme was voluntary, but more than 80% of farmers in the designated NSAs applied to join (MAFF 1991a).

1.3.2 N CODE OF PRACTICE

Increasing efforts are being made to educate farmers about the problems of NO_3-N pollution and to encourage codes of practice which lessen the risk of NO_3-N leaching. The basic code of practice that has been suggested to farmers is (*e.g.* Farman 1989):

- N fertiliser, manure or slurry should not be applied in the autumn;
- soil should not be left bare in the winter;

- sow winter crops early in the autumn;
- if a spring crop is to be grown, then grow a winter catch crop;
- avoid ploughing grassland in the autumn and delay to late winter if possible;
- avoid excessive applications of manure and slurry;
- adhere to recommended N rates and use split applications;
- do not use "insurance" dressings;
- ensure accurate calibration of equipment to minimise excessive applications.

The above recommendations are all contained in the MAFF *Code of Good Agricultural Practice for the Protection of Water* which is a Statutory Code under the 1989 Water Act. This means that although contravention will not itself give rise to liability, failure to comply with the Code could be taken into account in any legal proceedings (MAFF 1991c).

Whilst losses of fertiliser N (and soil-derived N) from the soil N cycle are a potential environmental and public health hazard, they also represent a major economic loss to the farmer. The Royal Society (1983) suggested that up to 326 000 tonnes of NO_3-N may be lost each year from UK agriculture by leaching alone. Assuming current N costs to be £0.36 kg-1 (SAC 1990), this is equivalent to about £1.2 million worth of potentially utilisable N lost each year. There is therefore also a considerable financial incentive for farmers to adopt the suggested code of practice in order to improve their utilisation of both fertiliser and soil-derived N.

1.3.3 AGRICULTURAL POLICY REFORM

Fertiliser N use has come under political scrutiny because of its contribution to the overproduction of many agricultural commodities, notably cereals, under the Common Agricultural Policy (CAP) of the EEC (Jollans 1985).

In 1987 alone the EEC spent £16 billion on agriculture, most of it on the purchase, storage and disposal of surplus production (Mackenzie 1988). A number of options have been considered in order to control the over-production of cereals. For example, the Nitrate

Co-ordination Group suggested that there was scope for developing a common policy on fertiliser N restriction to encompass both the problems of NO_3-N pollution and the CAP (DoE 1986). Such a policy might have included the imposition of fertiliser N quota's or taxation (*e.g.* Hartley 1986). However in 1988, in response to an EEC directive, the UK government introduced a 5 year voluntary set-aside scheme paying farmers prepared to take at least 20% of their land out of production of relevant arable crops (SAC 1989).

The amount of land currently out of production is about 3.5% of the total 3.3 million hectares of cereals land in the UK (MAFF 1991b). Set-aside land can be used either for rotational or permanent fallow, woodland or permitted non-agricultural use. It is difficult to assess the effect of the set-aside scheme upon agricultural N practice. Rotational set-aside can be used to reduce the N requirement of the following crop (SAC 1989), but experience in the USA (Raymond 1988), and increasingly in the UK, suggests that farmers are most likely to permanently set-aside their poorer land and intensify production on that which remains. According to MAFF (1990b) approximately 80% of those farmers registered on the scheme have taken the permanent fallow option and only 9% the rotational option.

1.3.4 DIFFERENT APPROACHES TO FARMING

In addition to the improved fertiliser and management practices cited above, some basic shifts in the pattern of agricultural production can be envisaged which might effect fertiliser N use. In particular, there is growing interest in low input agricultural systems which use "substantially lower levels of manufactured fertilisers, other agrochemicals, fuels and purchased concentrate feed per hectare or per livestock unit than is typical of current production systems in industrialised countries" (Wagstaff 1987). This description obviously covers a broad range of agricultural activities, including organic farming.

There are over 700 organic farmers in the UK with around 25,000 ha under full organic production (representing about 0.2% of all agricultural land) and a further 24,000 ha estimated to be under conversion (Redman 1991). The area under organic production in Scotland has doubled over the past 5 years and is currently estimated at between 2 400 and 3 000 ha (Daw *et al.* 1991).

The most widely accepted definition of organic farming is that adopted by the United States Department of Agriculture (Lampkin 1990):

> "Organic farming is a production system which avoids or largely excludes the use of synthetically compounded fertilisers, pesticides, growth regulators and livestock feed additives. To the maximum extent feasible, organic farming systems rely on crop rotations, crop residues, animal manures, legumes, green manures, off-farm organic wastes, mechanical cultivation, mineral bearing rocks, and aspects of biological pest control to maintain soil productivity and tilth, to supply plant nutrients and to control insects, weeds and other pests."

For a full and comprehensive account of organic farming principles and practice see Lampkin (1990).

1.4 RENEWED INTEREST IN THE USE OF LEGUMES

Interest in low input systems has focused attention upon legumes again and the use of symbiotic N_2 fixation as an alternative N source to fertilisers. Legumes can be used in an agricultural system as arable, forage or green manure crops :

a) Arable legumes are a combinable crop, grown primarily as a protein source for animal feeds. Since the introduction of EEC price support, field peas (*Pisum sativum*) and field beans (*Vicia faba*) have emerged as profitable break crops in arable rotations. There are no current figures on UK production, but according to Sprent (1982) about 150 000 ha of arable legumes were being grown annually in the UK, of which about 62% were planted with field peas, 30% with field beans and 8% with haricot beans (*Phaseolus spp.*);

b) Forage legumes are grown for cutting or grazing, either in pure stands (*e.g.* lucerne) or mixed leys (*e.g.* red clover/Italian ryegrass), or as a component of long term grassland (*e.g.* white clover/Perennial ryegrass). There were over 7.1 million ha of grassland in the UK in the early 1980s, the majority of which (70%) was long term *i.e.* over 5 years old (Cowling 1982). About 60% of this long term grassland had a clover content of 1-25%. The area of forage legumes grown in pure stands was relatively insignificant.

   With changes in the profitability of livestock production in the UK many farmers have used forage legumes such as white clover (*Trifolium repens*) to reduce fertiliser costs whilst still maintaining reasonable levels of output (Morrison *et al.* 1985). It has been suggested that the national UK level of grassland performance could, in principle, be achieved from productive clover swards without fertiliser N (Holmes 1976). Forage legumes are also a very important component of 'organic' farming rotations (Lampkin 1990). In organic farming systems in North America, forage legumes commonly occupy 25-50% of the area cultivated and are claimed to provide the majority of N requirements within the system (Power and Doran 1984);

c) Leguminous green manures are legume crops grown specifically for incorporation into the soil. Page (1922) wrote about the regular use of red or white clover as green manures in the potato growing districts of Ayrshire and the Lothians in the early 20th century. In more recent times however, the use of leguminous green manures in conventional agriculture has declined to insignificance (Parsons 1984), although they are an important part of many UK organic farming systems (Lampkin 1990). Depending upon local conditions, leguminous green manures are usually grown either as an over-wintering (*e.g.* red clover/ryegrass) or autumn (*e.g.* winter vetch) catch crop. Alternatively, they can be undersown (*e.g.* trefoil) in cereal crops (Schmid and Kläy 1982).

   Leguminous green manures may undergo a renaissance in conventional agriculture as a means of profitably managing set-aside land, since legumes can be included in the cover crop that must be maintained on the 'fallow' set-aside land (SAC 1989). American farmers, for example, were advised to use leguminous cover crops to manage set-aside land during the 1983 US Farm Acreage Reduction Program (Hicks *et al.* 1983).

1.4.1 ESTIMATES OF N_2 FIXATION

N_2 fixation is inherently variable depending upon the number of active nodules, their size and longevity, and the bacterial strains occupying them (Russell 1973). These factors in turn are affected by the complex interaction of legume species and cultivar, crop management and conditions of growth (notably water availability and soil nutrient status).

In an extensive review of North American work, La Rue and Patterson (1981) quoted estimates of N_2 fixation in the range of 10-100 kg N ha^{-1} for arable legumes and 100-250 kg N ha^{-1} for forage legumes. Cowling (1982) reported annual N_2 fixation estimates for forage legumes in the UK in the range of 74-280 kg ha^{-1} for mixtures of white clover and grass; 90-342 kg ha^{-1} for lucerne crops; and 219 kg ha^{-1} for red clover. In contrast to experimental plot results, however, Cowling (1982) claimed that the average N_2 fixation by forage legumes under agricultural conditions was only 11 kg N ha^{-1}. This low figure apparently reflects the poor management of forage legumes by U.K. farmers *e.g.* continued high fertiliser N applications to grass/clover swards and the preponderance of low (1-5%) clover contents.

Further estimates for N_2 fixation in field-grown crops relevant to the UK are summarised in Table 1.3. The methodology involved is reviewed in 2.2.

1.4.2 THE AGRONOMIC VALUE OF LEGUMES IN ROTATION

As already mentioned, it has been known since ancient times that legumes can exert a beneficial effect upon the growth of succeeding crops. This is most commonly attributed to the increased supply of N available to succeeding crops upon decomposition and mineralisation of the leguminous crop residues.

Dyke *et al.* (1977) found that the growth and incorporation of trefoil (*Lotus corniculatis* L.) as a green manure consistently increased barley yields (mean increase of 1.4 t ha^{-1}) during a long-term green manuring experiment. Crowley (1975) also found increases of up to 1.3 t ha^{-1} in winter wheat after red clover and in the USA, field peas significantly increased soil mineral N levels and winter wheat yields when compared to the effect of a preceding summer fallow or spring barley crop (Mahler and Auld 1989). Other researchers, however, have found that a large proportion of the N released from legumes is retained in the soil rather than being taken up by following crops (*e.g.* Müller and Sundman 1988).

Table 1.3 : N_2 fixation estimates for field-grown crops relevant to the UK (all estimates based upon ^{15}N dilution technique)

		N_2 fixation		
Crop:	Location:	**kg N ha^{-1}**	**%N_{dfa}**	Reference:
Red clover (*Trifolium pratense*)	UK	49.7 - 63.0*	-	Witty (1983)
Field peas (*Pisum sativa*)	UK	67.4 - 86.5*	-	Witty (1983)
	Denmark	102 - 215	44 - 64	Jensen (1986a)
	Denmark	244**	79	Jensen (1987)
Field beans (*Vicia faba*)	UK	166 - 181*	-	Witty (1983)
	Austria	79	74	Danso *et al.* (1987)
	Denmark	150 - 211**	60 - 70	Jensen (1986a)
	USA	125 - 144*	71 - 84	Wagner and Zapata (1982)
French beans (*Phaseolus vulgaris*)	UK	84.1 - 100.0*	-	Witty (1983)

N.B. : * estimates made in same season using different reference crops;
** estimates made over 3 consecutive seasons.

The problem with attempting to predict the agronomic value of legumes in rotation is that the processes controlling the release and fate of legume-derived N are very complex.

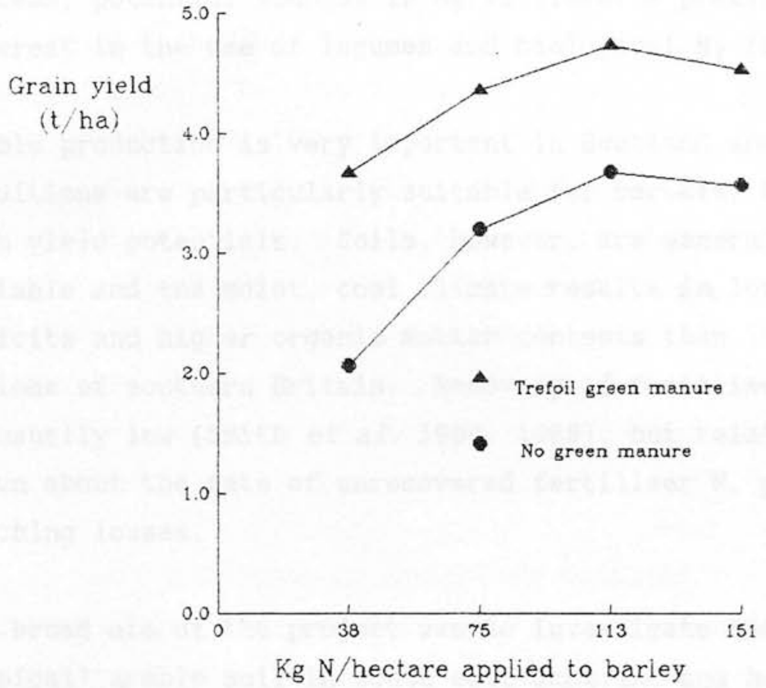
Availability of legume N is tied to the general patterns of organic matter decomposition in the soil (*e.g.* Frankenberger and Abdelmagid 1985). Thus climate, cultivations and other management practices have a potentially more complex effect upon N availability in systems exploiting legume N than in those using fertiliser N inputs. Microbial turnover of legume N may cause net release to the crop to be insignificant (Ladd *et al.* 1981) and rapidly decomposing fresh legume residues may be conducive to high rates of denitrification (Smid and Beauchamp 1976). Patterns of legume N release may lead to high NO_3-N leaching losses (Adams and Pattinson 1985), in other cases it may not (Müller 1987).

The beneficial effect of legumes upon succeeding crops may not be solely due to increased N availability, but may also include so-called 'rotation effects' *i.e.* yield benefits derived simply from changing the sequence of crops in rotation.

The association of a 'rotation effect' with legumes is unclear. In 10 years of rotation studies Schrader *et al.* (1966) found no effect of legumes on successive crops other than to provide N. Smith *et al.* (1987) cited numerous authors whose work indicates that legumes provide benefits beyond simply N supply, but emphasised that these benefits were not observed consistently. Dyke *et al.* (1977), for example, found that under some circumstances the incorporation of a leguminous green manure caused a yield increase in barley which could not be matched by any application of fertiliser N (Figure 1.2). Smith *et al.* (1977) concluded that the only valid general statement on legume 'rotation effects' is that whereas the N contribution from legumes is almost always observed, additional benefits are not.

The value of legumes in rotation will also depend very much upon their management. When arable legumes are harvested up to 75% of the total N content of the crop can be removed in the grain,

Figure 1.2: Yield of barley with and without a previous green manure crop (taken from Dyke *et al.* 1977)



significantly limiting the contribution of legume N to the nutrition of succeeding crops (Goh and Haynes 1986). This means that even when levels of N_2 fixation are relatively high, there can be more soil-derived N lost in the grain than there is atmospherically-derived remaining in crop residues (Heichel and Barnes 1984); thereby leading to the phenomenon of a legume reducing soil N status rather than increasing it.

When legume leys are cut for conservation a large proportion of the crop N is also lost and the contribution of legume N to subsequent crops can be limited (Goh and Haynes 1986). Problems of soil N depletion, as with arable legumes, have also been reported in conserved legume leys (Rice 1980). When legume leys are grazed however, most of the crop N is returned to the soil via dung and urine and the contribution of legume N to following crops is greater (Goh and Haynes 1986).

1.5 RESEARCH AIMS AND OBJECTIVES

This research project was established and conducted against the preceding background of concern over N 'leakage' from agricultural systems, potential changes in agricultural N practice and renewed interest in the use of legumes and biological N_2 fixation.

Arable production is very important in Scotland and environmental conditions are particularly suitable for cereals, often resulting in high yield potentials. Soils, however, are generally shallow and variable and the moist, cool climate results in lower soil moisture deficits and higher organic matter contents than in the arable regions of southern Britain. Recovery of fertiliser N on these soils is usually low (Smith *et al.* 1984, 1988), but relatively little is known about the fate of unrecovered fertiliser N, particularly NO_3-N leaching losses.

The broad aim of the project was to investigate the N dynamics of a 'typical' arable soil in south-east Scotland and how these are modified by:

i) reducing fertiliser N application;
ii) replacing the fertiliser N with a leguminous source of organic N;
iii) growing a winter cover crop.

More specific research objectives pursued in the course of the investigation were:

a) the direct measurement and comparison of NO_3-N leaching losses from the different N treatments;
b) a study of the use of a legume as a source of N for crop growth under local conditions, including the estimation of biological N_2 fixation and mineralisation of leguminous crop residues;
c) the preparation of detailed N balances for the different N treatments.

All experimentation was field-based, with the main emphasis upon the direct measurement of NO_3-N leaching losses from hydrologically isolated plots, complemented by routine measurements of crop N uptake, soil mineral N, atmospheric N deposition and N_2O flux.

1.5.1 THE USE OF HYDROLOGICALLY ISOLATED PLOTS

Methods available for the direct measurement of leaching losses from arable land include the collection and sampling of drainage discharge from isolated catchments, field drains, hydrologically isolated plots and lysimeters (reviewed in 2.1). The two latter methods have both been used effectively as the basis of successful N cycle studies (*e.g.* Dowdell 1984), but hydrologically isolated plots were specifically chosen for this project because of their apparent suitability for measuring leaching losses under local soil and topographical conditions.

Much of the arable land in south-east Scotland has a gently undulating topography derived from the mantle of glacial-till that covers the region. Individual sites therefore tend to be sloping and have relatively impermeable, compact subsoils. A "perched" water table often exists at the interface between the plough layer and the subsoil, and the movement of saturated flow and solutes tends to occur as downslope lateral flow (Figure 1.3). Little information exists on NO_3-N leaching from such soils, indeed it is often assumed that leaching is minimal because of the heavy texture and imperfect drainage.

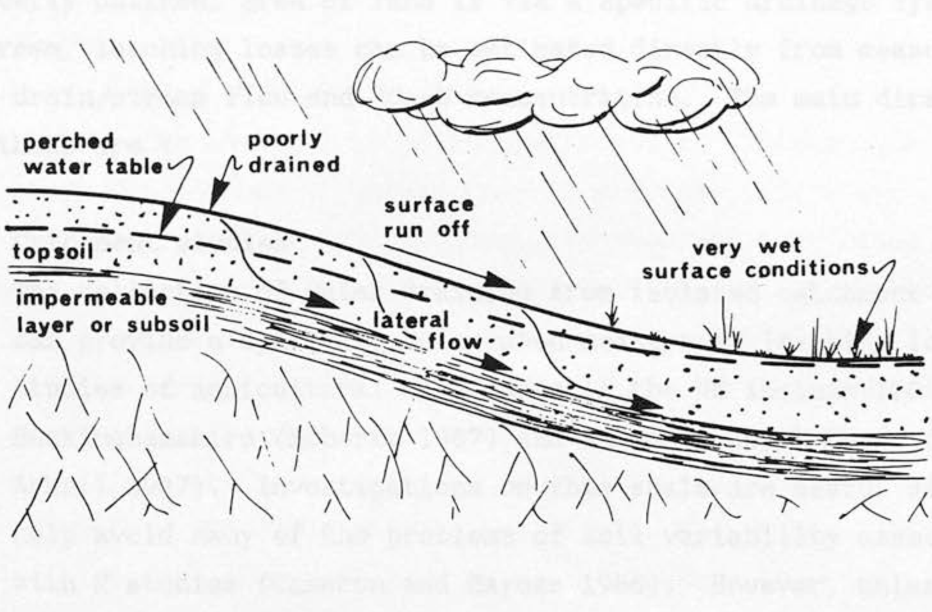
It was proposed to take advantage of downslope lateral flow in order to establish the hydrologically isolated plots from which NO_3-N leaching losses would be measured. On a sloping site, the isolation of a plot on 3 sides should enable the lateral flow from the plot to be intercepted along the bottom edge of the plot ready for drainflow measurement and sampling.

The plots were intended to be similar in effect to those used in the Brimstone Farm drainage experiment in Oxfordshire (Cannell *et al.* 1984). It was hoped that the occurrence of a deep, heavily compacted subsoil would limit the problem of plot drainage gain by groundwater inflow experienced at Brimstone (Harris *et al.* 1984), as well as the losses of plot drainage by deep percolation experienced in other work (*e.g.* Bergström 1987).

In order to test the efficacy of plot isolation in local soils, a single hydrologically isolated pilot plot was established and rainfall recovery data collected for 5 months over the 1986-87 winter (Chapter 3). Work was also carried out on the development of flow measurement and water sampling techniques, and the collection of some basic winter leaching data.

The results obtained from the pilot plot were acceptable and during the spring and summer of 1987 work proceeded with the establishment of a further 8 hydrologically isolated plots at a suitable site. The main experimental period commenced in September 1987 (Chapter 4).

Figure 1.3: Occurrence of lateral flow in soils with impermeable subsoils (taken from SAC 1983)



CHAPTER 2: LITERATURE REVIEW

The availability of literature on N cycling, and its study, in arable soils is overwhelming. To attempt even a broad review is neither possible, nor desirable, in the context of this thesis. Instead, it is proposed to limit the following literature review to two specific aspects of the methodology used in this research project: the field study of NO_3-N leaching losses and the estimation of symbiotic N_2 fixation under field conditions.

2.1 APPROACHES TO THE FIELD STUDY OF LEACHING LOSSES

NO_3-N leaching losses are not easy to study in the field. However many attempts have been made and a variety of different approaches have been used. These can be broadly classified as direct and indirect, and are briefly reviewed here: firstly as background to the methodology used for the leaching studies in this work and secondly to allow an appreciation of the source and limitation of leaching results reported and discussed in this thesis.

2.1.1 DIRECT METHODS

Where the main pathway of water movement from an isolated, and clearly defined, area of land is via a specific drainage system or stream, leaching losses can be estimated directly from measurements of drain/stream flow and NO_3-N concentration. The main direct methods are :

A. Catchment studies

The collection of water draining from isolated catchment areas can provide a spatially integrated measure of leaching loss. Studies of agricultural catchments in the UK include 170 ha in Buckinghamshire (Roberts 1987) and 94 ha in South Devon (Burt and Arkell 1987). Investigations on this scale are useful since they help avoid many of the problems of soil variability associated with N studies (Cameron and Haynes 1986). However, unless land-use and soil management is uniform within a catchment it is difficult to relate specific agricultural practices to NO_3-N leaching losses. This can be overcome to a certain extent by sampling streams and drain outfalls from contrasting areas within the catchment (*e.g.* Burt and Arkell 1987), although this is likely to be limited by the poor estimation of drainage discharge from poorly isolated sub-catchment areas.

There are large variations in the relationship between NO_3-N losses and soil management reported in catchment studies, and this is attributed to differences in the experimental conditions of individual studies (Burwell *et al.* 1976). For example, Burt and Arkell (1987) found that topography can have a major influence upon the pattern of NO_3-N loss from a catchment with valley side slopes particularly losing a lot of NO_3-N per unit area.

B. Field drain studies

The relationship between a specific agricultural practice and NO_3-N leaching can be investigated by collecting water from field drains. Reported field drain studies include investigations of the chemical composition of water from drains at Woburn and Saxmundham Experimental Stations (Williams 1971); NO_3-N leaching from intensively managed grassland at Jealott's Hill (Hood 1976); and NO_3-N leaching from arable soils under different cropping systems in Sweden (Gustafson 1987).

An important limitation of field drain studies is that water samples for analysis are often collected at a fixed time interval (*e.g.* twice a month) rather than on a volume flow basis (Cameron and Haynes 1986). Flow proportional sampling techniques have been used on field drains (*e.g.* Hood 1976), but it has been questioned whether they are justified (Williams 1970). Field drains only intercept a proportion of the water flow down through the profile. This proportion is always unknown, probably changing seasonally as the capacity of subsoil drainage channels varies (Williams 1970, Wild and Cameron 1980). Therefore the water sampled collected may never be representative, or even consistently mis-representative, of the total field drainage.

C. Plot drain studies

The oldest plot drain study recorded in the UK was made at Rothamsted. Lawes *et al.* (1882) (cited by Addiscott 1988) used the drainage system installed on Broadbalk field to measure plant nutrient losses from experimental plots under continuous wheat. A number of more recent plot drain studies have also been made, often utilising plots that were established principally for field drainage experiments such as at Brimstone Farm, Oxfordshire (Cannell et al. 1984), Cockle Park, Northumberland (Armstrong *et al.* 1980) and North Wyke, Devon (Scholefield *et al.* 1988). Plot drain studies can be considered distinct from field drain studies in that the areas drained are considerably smaller and there is usually some element of hydrological isolation to prevent the movement of water into and out of plots. An important advantage of plot studies is obviously that NO_3-N leaching from various experimental treatments can be measured at the same site and under similar physical conditions.

Plots are usually large enough for normal agricultural practices to be conducted on them. Scholefield *et al.* (1988) reported the measurement of leaching losses from 1 ha grassland plots (receiving differential N rates) that were grazed by beef cattle, and Goss *et al.* (1988) compared NO_3-N losses from direct and conventionally drilled plots. Bergström (1987) also reported the measurement of leaching losses following the ploughing of grass and lucerne leys established in plots in Sweden.

Most experimental sites are sloping to assist the collection of drainage water from plots. For example, at Brimstone Farm drainage from the sloping plots were collected in 1.0 m deep trenches, laid with clayware pipes and filled to the surface with permeable backfill (Cannell *et al.* 1984). Sloping sites can also be exploited for hydrological isolation of plots, with the exact approach to plot isolation varying according to experimental requirements and specific site characteristics.

Burke (1975) simply used an open isolation channel to prevent water moving into plots established on a blanket peat bog in

Ireland. At many sites, however, 1.0 - 1.3 m deep interception drains with permeable backfill have been installed across the top edge of plots to prevent the influx of water from upslope (*e.g.* Armstrong *et al.* 1980, Cannell *et al.* 1984).

In some studies the isolation of the individual plots is relatively simple. Armstrong *et al.* (1984) just installed a series of 30 cm deep drains (backfilled to the surface) on the plot boundaries which intercepted surface run-off leaving the plots at North Wyke. Individual plot isolation at Cockle Park (Armstrong *et al.* 1980) and Brimstone Farm (Cannell *et al.* 1984) was rather more sophisticated using continuous vertical polythene barriers installed to at least 1 m depth and brought to the surface. Particular care may also be taken to render old drains ineffective by lifting and plugging with bentonite at the point they crossed new drains (*e.g.* Armstrong *et al.* 1984, Cannell *et al.* 1984).

A potential limitation of plot drain studies can be the efficacy of plot drainage and hydrological isolation, and as a result they may not always give an accurate picture of water and NO_3-N losses from the soil profile (examples are given in 6.1).

D. Lysimeter studies

A lot of the available information on NO_3-N leaching is based upon lysimeter experiments. Lysimeters have been used for at least 250 years in studies of water movement through soil (Low and Armitage 1970) and are of two main types :

a) "monolith" lysimeters - using undisturbed soil blocks, either walled-in by masonry (*e.g.* Hendrick 1921) or fitted into suitable containers (*e.g.* Low and Armitage 1970, Dowdell *et al.* 1984);

b) "filled" lysimeters - consisting of tanks, lined pits or other suitable containers filled with disturbed soil (*e.g.* Bergström 1987, Bertilsson 1988).

Lysimeters allow quantitative measurements of NO_3-N leaching from a defined soil volume, and have the advantage that all percolating water reaching the bottom of a lysimeter is measured and sampled. Barraclough *et al.* (1984) and Bergström (1987) found, for example, that measurements of drainage, and consequently NO_3-N leaching, were consistently higher from field lysimeters than from drained plots at the same site. Lysimeters have been used in N balance studies (*e.g.* Dowdell and Webster 1984) and legume N studies (*e.g.* Müller 1987), offering the advantages of a confined microplot (*e.g.* for detailed ^{15}N work) with the facility to measure NO_3-N leaching.

The most serious drawbacks with lysimeters are:

1. Crop growth in lysimeters may not be typical of growth in the field, notably because of the effect of soil containment on root proliferation, aeration, temperature and moisture conditions (Bergström 1987);
2. Container edge effects, due to soil shrinkage, may lead to increased soil aeration and preferential drainage pathways (Wild and Cameron 1980);
3. The risk of increased moisture content, or accumulation of drainage water in the lysimeter base, may lead to anaerobic conditions and denitrification (Harmsen and Kolenbrander 1965).

"Filled" lysimeters are easier to construct than "monolith" lysimeters, but are further limited by soil disturbance during installation leading to:

i) changes in unsaturated water flow, for example, Bergström (1987) reported that "filled" lysimeters display more rapid water movement and higher drainage volumes than "monolith" lysimeters;
ii) increased soil aeration and the stimulation of N mineralisation (Harmsen and Kolenbrander 1965).

Tension lysimeters have been used in some N cycle studies (Keeney 1986). These have no walls, and are based upon the placement of a suction plate horizontally below the soil surface. Tension lysimeters are of limited applicability - extensive excavation is required to install them (Keeney 1986) and, since the volume of soil contributing to the collected water is unknown, it is impossible to make absolute leaching measurements (Low and Armitage 1970).

2.1.2 INDIRECT METHODS

If drainage from a soil is such that it cannot easily be sampled, as in naturally free draining soils overlaying a porous parent material (*e.g.* calcareous soils), leaching losses must be measured indirectly or estimated in some other way. The main indirect methods are :

A. Field soil sampling

The simplest approach to estimating NO_3-N leaching is to take soil samples and measure the total NO_3-N content (kg N ha^{-1}) of a soil profile before and after any leaching is expected, and then regard any decrease in NO_3-N content as a leaching loss (*e.g.* Barraclough *et al.* 1983).

This is a reasonably accurate method of estimating NO_3-N leaching, but there are some important sources of error. Soil NO_3-N content can show very high spatial variability and sufficient soil samples must be taken to account for this (Keeney 1986). There is also considerable temporal variation due to soil N transformations. If NO_3-N levels increase due to the occurrence of mineralisation between sampling times, leaching will be underestimated; whilst if NO_3-N is removed by crop uptake, immobilisation or denitrification, leaching will be overestimated. These errors can be reduced by ensuring that sampling times are chosen well into the winter period when soil temperatures are low and biological N transformations are minimal (Powlson 1988), whilst at other times of the year crop N uptake can be accounted for (*e.g.* Adams and Pattinson 1985).

Cameron and Wild (1984) used a modification of this approach to determine the potential aquifer pollution arising from the ploughing of temporary grassland. They took soil samples to a depth of at least 165 cm and assumed that there were no biological N transformations active below 90 cm. Any observed increase in NO_3-N content (kg ha^{-1}) from 90-165 cm depth was therefore attributed to leaching from the soil above.

An alternative sampling method is to use ceramic suction cups, installed at one or more depths, to collect soil water (*e.g.* Groffman *et al.* 1987). The combination of soil water NO_3-N concentration and estimated drainage flux can then be used to calculate NO_3-N loss (kg ha^{-1}). Suction cups are, however, limited by the following factors:

i) a disproportionate amount of water is removed from large soil pores (containing water at low tension) and, since NO_3-N concentrations are likely to vary with pore size, collected samples are prone to bias;

ii) soil disturbance during installation may lead to erroneous results, notably mineralisation;

iii) drainage fluxes can be difficult to calculate (Keeney 1986, Powlson 1988).

B. Borehole sampling

Deep cores (up to 200 m deep) taken from unsaturated zones above aquifers can be used to sample interstitial water from porous rock types (*e.g.* Cretaceous Chalk and Triassic Limestone) and can provide useful information on the potential NO_3-N pollution of underground drinking water supplies (Foster *et al.* 1982). Studies have been mainly concerned with relating the vertical distribution of NO_3-N (mg l-1) in pore water to general agricultural practice (*e.g.* Young and Gray 1978), rather than quantifying the NO_3-N loss (kg ha^{-1}) from specific practices.

2.2 METHODS OF ESTIMATING N_2 FIXATION UNDER FIELD CONDITIONS

Over the last 10-15 years considerable research has been directed towards enhancing biological N_2 fixation in legumes *e.g.* selection of superior plant and *Rhizobium* genotypes; refinement of inoculation technology; and improvement of management practices (Smith and Knight 1984). Much of this work has been founded upon the use of pot-grown legumes, often under glasshouse conditions, but there have also been advances in the development and implementation of techniques for estimating N_2 fixation in the field. The range of available field techniques are reviewed here.

2.2.1 NITROGEN ACCUMULATION

The simplest estimate of N_2 fixation is the total N accumulation of the legume crop. This is based upon the intuitive assumption that the legume derives all of its N content via symbiotic fixation. There is however no published evidence that this is ever the case under field conditions, even on soils in which mineral N levels are artificially reduced (LaRue and Patterson 1981). For example, Kohl *et al.* (1980) reduced mineral N levels in a soil by the addition of 34 t ha^{-1} of high C:N ratio maize residues, and a subsequent crop of soyabeans still obtained 47% of its N content from the soil.

A closer approximation to N_2 fixation may be achieved by analysing changes in soil mineral N as well as that removed in the crop. However, this approach is still limited by the need for deep soil sampling (legume roots may reach a depth of up to 3 m) and the effect of other N cycle processes upon soil mineral N levels (LaRue and Patterson 1981).

2.2.2 DIFFERENCE METHODS

A modification of the N accumulation technique is to estimate the contribution of soil N to the total N yield of the legume through the use of a non-fixing reference crop (Rennie 1984), such that:

$$N_2 \text{ fixed} = \text{N yield (legume)} - \text{N yield (reference crop)} \qquad (1)$$

$$\%N_{dfa} = \frac{\text{N yield (legume)} - \text{N yield (reference crop)}}{\text{N yield (legume)}} \times 100 \qquad (2)$$

where: $\%N_{dfa}$ is the percent crop N derived from the atmosphere.

This procedure is often referred to as the "N balance" or "difference" method, and assumes equal uptake of soil N by the legume and reference crop. However, better N nutrition in an actively fixing crop may improve rooting and increase soil N uptake by the legume (Rennie 1984), thereby leading to an over-estimation of N_2 fixation. For example, Wagner and Zapata (1982) measured a higher uptake of fertiliser ^{15}N in the field by nodulated soyabeans than non-nodulated plants.

In pasture systems where forage legumes and grasses are grown together in mixed stands, estimation of the N transfer between the legumes and grasses is often used to indirectly estimate N_2 fixation (LaRue and Patterson 1981). If the yield of the grass in the mixed stand is compared to the yield response of the grass in a pure stand with different fertiliser N applications, then the transfer of legume N to the grass can be calculated as a "fertiliser N equivalence" (*e.g.* Chestnutt 1972).

2.2.3 ACETYLENE REDUCTION

The acetylene reduction assay (ARA) of Hardy *et al.* (1968) is a useful technique for determining potential nitrogenase activity in field-grown legumes (*e.g.* Rice 1980). Nitrogenase not only reduces N_2 to NH_3 but also acetylene (C_2H_2) to ethylene (C_2H_4), and these latter two gases can be rapidly and accurately analysed on a gas chromatograph. Typically, ARA involves the incubation of freshly excised legume roots in a chamber with 1-20% C_2H_2 for 30-120 minutes, followed by analysis of the gas phase for C_2H_4.

ARA has been used for estimating seasonal N_2 fixation (*e.g.* Goh *et al.* 1978), but this is a questionable application of the technique:

1. There are considerable problems in obtaining representative root samples for the assay, particularly where field variability is high (Goh *et al.* 1978);

2. ARA is only a short-term kinetic measurement and the existence of diurnal and seasonal variations in N_2 fixation makes extrapolation to total N_2 fixed over a growing season difficult unless very frequent ARA measurements are made (Rennie *et al.* 1978);
3. Estimation of N_2 fixation by ARA relies upon the use of a C_2H_2 reduced:N_2 fixed conversion factor (LaRue and Patterson 1981). This is rarely determined in fixation studies and a theoretical conversion factor of 3:1 is commonly used instead. The actual ratio of C_2H_2 reduced:N_2 fixed can however vary widely according to prevailing environmental conditions (*e.g.* light, temperature, pO_2 and moisture) such that major errors are likely if: a) the correction factor is not validated for the specific conditions of the ARA, and b) the conditions of the ARA do not exactly match those in the field (Bergersen 1970).

2.2.4 ISOTOPIC METHODS

The stable isotope ^{15}N has both been used as a definitive test to prove the occurrence of N_2 fixation and for the estimation of total seasonal fixation (Rennie *et al.* 1978).

If a potential N_2 fixing system is exposed to $^{15}N_2$ for a short period, and $^{15}NH_3$ or derivatives thereof are found in the sample, then N_2 fixation can be assumed to have occurred. But the use of $^{15}N_2$ is again only a short-term kinetic measurement like ARA, and as such is subject to similar limitations in the estimation of seasonal N_2 fixation.

Probably the most useful method available for estimating seasonal N_2 fixation in field-grown legumes is the ^{15}N isotope dilution technique, although the absolute validity of the technique is still debatable (Chalk 1985, Danso 1986). The technique was first proposed by McAuliffe *et al.* (1958) and is based upon the differential dilution of ^{15}N-labelled fertiliser in a fixing legume and non-fixing reference crop. The reference crop has 2 possible sources of N, soil and fertiliser, whilst the legume has a third source, the atmosphere. Assuming that the legume and reference crop assimilate soil and fertiliser N in the same proportion over the growing season (but not necessarily in the same quantities), any reduction in ^{15}N enrichment in the legume is due to dilution by atmospherically-derived $^{14}N_2$ (Rennie and Rennie 1983). The following equations can then be used:

$$\%N_{dfa} = \left[1 - \frac{\text{atom\% } ^{15}\text{N excess (legume)}}{\text{atom\% } ^{15}\text{N excess (reference crop)}} \right] \times 100 \quad (3)$$

$$N_2 \text{ fixed} = \frac{\%N_{dfa}}{100} \times \text{total N (legume)} \quad (4)$$

The N content of the legume can be further investigated by:

$$\%N_{dff} = \frac{\text{atom\% } ^{15}\text{N excess (plant)}}{\text{atom\% } ^{15}\text{N excess (fertiliser)}} \times 100 \quad (5)$$

$$\%N_{dfs} = 100 - (\%N_{dff} + \%N_{dfa}) \quad (6)$$

where: $\%N_{dff}$ and $\%N_{dfs}$ are the percent crop N derived from fertiliser and soil respectively.

Most N_2 fixation studies are carried out at low fertiliser N levels in order to avoid inhibition of the N_2 fixation under study, but as a result of this the reference crop may suffer conditions of N deficiency (Chalk 1985). Fried and Broeshart (1975) therefore proposed the "A" value modification, advocating a larger fertiliser N application to the reference crop to allow it to be grown with an adequate N supply.

The "A" value is the amount of an available nutrient in the soil expressed in terms of a fertiliser standard (Fried and Dean 1952). When used as a measure of available N the "A" value is expressed (Rennie and Rennie 1983) as:

$$\text{"A" value} = \frac{(100 - \%N_{dff})}{\%N_{dff}} \times \text{rate of fertiliser N applied} \quad (7)$$

N_2 fixed is then calculated (Fried and Broeshart 1975) as:

$$N_2 \text{ fixed} = \left[\text{"A" (legume)} - \text{"A" (reference)} \right] \times \frac{\text{FUE (legume)}}{100} \qquad (8)$$

where: FUE = fertiliser use efficiency

$$= \frac{\%N_{dff} \times \text{total N (legume)}}{\text{rate of fertiliser N applied}} \qquad (9)$$

When equal amounts of labelled N are applied to the fixing and reference plants, the "A" value method is mathematically identical to the classical isotope dilution method expressed in equations 3 and 4 (Fried and Middelboe 1977).

The "A" value modification has been widely used (Chalk 1985), but its validity depends upon the "A" value being independent of fertiliser N rate. Several studies have shown that the "A" values for reference crops vary with increasing rates of N addition, although the results have not been consistent. Deibert *et al.* (1979) found that "A" values for non-nodulated soyabeans increased with the N rate applied at planting, and this was attributed to a soil N priming or root extension effect. In contrast, Boddey *et al.* (1983) found that reference crop "A" values decreased with increasing N rate, indicating a greater uptake of fertiliser N relative to soil N.

The use of the isotope dilution technique to estimate N_2 fixation is totally dependent upon the comparison of the legume and reference crop. A number of studies have shown significant differences in the estimation of N_2 fixed when using different reference crops (Wagner and Zapata 1982, Witty 1983, Fried *et al.* 1983), suggesting that accurate use of the isotope dilution method depends very much upon the choice of an appropriate reference crop.

There are a number of different types of reference crop. A non-nodulating cultivar has frequently been used as a reference for soyabean studies, whilst in soils lacking indigenous Rhizobium species uninoculated legumes have been used (Chalk 1985). The most convenient reference crops available however are non-legumes. For example, barley (*Hordeum vulgare*) has commonly been used in arable legume studies (*e.g.* Jensen 1986a) and ryegrass (*Lolium mutiflorum*) in forage legume studies (*e.g.* Reichardt *et al.* 1987).

According to Witty (1983) the ideal legume-reference crop combination should have a) similar N uptake patterns, and b) similar rooting patterns.

1. **N uptake patterns**

   The need for similar N uptake patterns in the legume-reference crop combination is generally accepted by most researchers (Chalk 1985) and is related to the problems of achieving a stable ^{15}N enrichment of the soil mineral N pool.

   The most common method of applying fertiliser ^{15}N in N_2 fixation studies is to spray an aqueous solution onto designated sub-plots before the legume and reference crops emerge (Rennie 1986). Following application of the ^{15}N fertiliser, enrichment of the soil mineral N pool declines rapidly due to crop uptake, loss and immobilisation of the added ^{15}N, combined with the continued release of inorganic ^{14}N by mineralisation (Witty 1983). If the N uptake patterns of the legume and reference crop are not matched, then differences in plant isotopic ratio at harvest are not solely due to dilution of ^{15}N by atmospheric $^{14}N_2$, but also due to differences in ^{15}N:^{14}N uptake from the soil mineral N pool.

   In matching a legume and reference crop it is therefore not the quantity of N uptake that is important, but rather the time of N uptake relative to the decline in soil ^{15}N enrichment (Rennie and Rennie 1983).

2. **Rooting patterns**

   The need for similar rooting patterns in the legume-reference crop combination is debatable. Witty (1983) maintained that if the legume and reference crop are to utilise the same mineral N pool, then either ^{15}N-labelled fertiliser distribution must be entirely uniform with depth, or the rooting patterns must be the same. Other researchers (*e.g.* Rennie 1984) suggested that rooting pattern is not important as long as the plough layer (containing most of the fertiliser and soil N) is adequately explored by roots.

The issue is complicated by the different behaviour of $^{15}NH_4$ and $^{15}NO_3$ fertiliser in the soil (Chalk 1985). Whilst $^{15}NH_4$ remains predominantly in the plough layer, $^{15}NO_3$ may be leached below the root zone of shallow rooted crops and thus not be equally accessible to legume and reference crops differing in rooting pattern. For example, Ledgard *et al.* (1985) noted that estimates of $\%N_{dfa}$ for clover grown with ryegrass (as reference) were dependent upon the volume of water applied with the $K^{15}NO_3$ fertiliser. Increasing the volume of water from 2 to 10mm leached the $^{15}NO_3$ further down the soil profile, and induced a greater ^{15}N uptake by the ryegrass, presumably because it was deeper rooting.

Witty (1983) suggested that the best approach to selecting a suitable reference crop (at least in initial experiments) may be the use of a range of crops, so that researchers can gain some feel for which is most appropriate. The same author further suggested that whilst a perfect matching of legume and reference crop is seldom likely to be possible, the error incurred by a mismatch can be reduced by using a labelling technique that produces a more stable ^{15}N enrichment of the soil mineral N pool.

A number of alternative labelling techniques have been used, although it should be noted that in many cases their primary objective has been to avoid the inhibitory effects of high levels of mineral N rather than to encourage a stable enrichment. Simple variations of the common fertiliser N application technique have included the multiple addition of small amounts of highly labelled fertiliser N throughout the growing season (Chalk 1985), and the use of slow-release ^{15}N fertiliser formulations *e.g.* Witty (1983) applied gypsum-pelleted $K^{15}NO_3$.

Several investigators have labelled the soil organic N pool by the addition of ^{15}N enriched plant material. This has two specific advantages: (i) it provides a slow-release ^{15}N label; and (ii) residues from previous ^{15}N experiments can be re-used (Rennie 1986). For example, Fried *et al.* (1983) added labelled plant material by turning in residues from an $(^{15}NH_4)_2SO_4$-fertilised spring wheat crop in the autumn preceding the establishment of legume plots the following year. Such long-term planning of experiments may be a problem, but relatively stable soil mineral N enrichments can be achieved 6 - 18 months after incorporating the labelled material (Chalk 1985). It is apparently not necessary to have a high ^{15}N label in the plant material incorporated, but the higher the enrichment then the better the experimental precision (Rennie 1986).

The soil organic N pool has also been labelled by the addition of $(^{15}NH_4)_2SO_4$ and an available C source, such as sucrose, to stimulate microbial immobilisation of the fertiliser N (Chalk 1985). A major disadvantage with this method is that the resultant changes in microbial activity may be untypical of normal field conditions (Fried *et al.* 1983). In particular, if too much available C is added then the immobilisation of indigenous soil N can result in higher levels of N_2 fixation than would have occurred in the field. For example, Wagner and Zapata (1982) found that the percentage N derived from the atmosphere in soyabeans was increased from 30 to 80% following the addition of sucrose.

Biological processes in the soil tend to favour the lighter ^{14}N isotope over the heavier ^{15}N, and this results in the soil mineral N pool usually having a slightly higher ^{15}N abundance than the atmosphere (Rennie and Rennie 1983). It has been claimed that this natural ^{15}N enrichment can be used to accurately estimate $\%N_{dfa}$ without the need for ^{15}N addition, thus largely avoiding the problems associated with ^{15}N labelling (Rennie 1986). However, there are several constraints to the use of natural abundance techniques. N_2 fixation may be susceptible to isotopic discrimination, such that under certain ecological and physiological conditions fixation of ^{14}N is favoured (LaRue and Patterson 1981). Whilst there is little evidence to suggest that significant isotopic discrimination does occur during N_2 fixation (Rennie 1978), it should be accounted for when working at levels of natural ^{15}N abundance (Rennie and Rennie 1983). Another disadvantage is that although the natural enrichment of the soil mineral N pool is stable, it may not be distributed uniformly (LaRue and Patterson 1981). Indeed it has been suggested that the spatial variability in natural enrichment may be of the same order of magnitude as the difference between soil and atmospheric N (Rennie and Rennie 1983). Because of these constraints, the use of natural abundance techniques is generally considered to only give qualitative or semi-quantitative information.

The accuracy and precision of the isotope dilution technique depends upon a number of other factors besides just the reference crop and labelling technique. These include:

A. **Site heterogeneity**
Reichardt *et al.* (1987) concluded that the sensitivity of the isotope dilution technique can be greatly improved by selecting fairly homogeneous experimental sites. Although heterogeneous sites are considered acceptable if the reference and legume crops are sited close together (Fried et al 1983, Reichardt *et al.* 1987). Errors associated with soil spatial variability or mismatching of legume and reference crops apparently become less as the level of N_2 fixation increases (Danso 1986, Reichardt *et al.* 1987).

Pasture systems would therefore appear to be ideal for N_2 fixation measurements on heterogeneous soils, because the grass (reference) and legume are intimately mixed. With arable legumes, the reference can be inter-cropped in alternate rows (Chalk 1985).

A potential experimental difficulty in having the reference and legume crops too closely associated is that N transfer can occur from the legume to the reference thus resulting in an underestimation of N_2 fixation (Fried *et al.* 1983).

B. ^{15}N labelling of plant tissue

Crops grown on ^{15}N enriched soils are not uniformly labelled (Chalk 1985). For example with legumes, the leaves are frequently more enriched than the stems, and the pods and seeds are least enriched. This is related to a number of factors including, generally the interaction of crop development with changes in ^{15}N enrichment of the soil mineral N pool, and more specifically with legumes, the interaction of crop development with changes in the proportions of soil and atmospheric N assimilated.

Non-uniform labelling of the legume and reference crops means that the sensitivity of the isotope dilution technique is greatly affected by the selection of the plant parts used for the calculation of $\%N_{dfa}$ (Rennie *et al.* 1978). Proper plant sampling is therefore crucial. Fried *et al.* (1983) suggested that plants should be stripped into recognisable fractions such as seeds, pods, leaves and straw. These should be sub-sampled and analysed for total N content and ^{15}N enrichment, then the results should be combined to obtain a weighted atom% ^{15}N excess on a whole-plant basis, which can be used to calculate $\%N_{dfa}$. The harvested proportion of the crop used in N_2 fixation studies usually consists of the aerial tissue (Chalk 1985). Bergersen and Turner (1983) claimed that this was quite adequate when $\%N_{dfa}$ was greater than 50%, and the considerable effort of root collection could be avoided. However, Bergersen and Turner (1983) also stressed that the total N content of roots represents a considerable proportion of the total legume N yield, and cannot be ignored when estimating the amount of N_2 fixed.

C. Seedborne N

In addition to soil, fertiliser and atmospherically-derived N, legumes and reference crops also derive some of their N content from seed-borne N. This seed-borne N is generally ignored when calculating N_2 fixation (Chalk 1985), but Jensen *et al.* (1985) concluded that it is important to correct N_2 fixation estimates for seed-borne N particularly when there are large differences in the seed N content of the legume and reference crop. Witty and Ritz (1984) however felt that this was only likely to be necessary where the proportion of seed N present in the legume was high (*e.g.* during the early stages of legume development).

CHAPTER 3 : BOGHALL FARM PILOT PLOT STUDY

A single hydrologically isolated pilot plot was established on the Edinburgh University's Bush Estate at Boghall Farm, near Easter Howgate, Midlothian (O.S. grid reference NT245652) in July 1986 prior to the start of the main project. Work was subsequently conducted on the plot from November 1986 to early April 1987.

The main objectives of the pilot plot were to:

1. Assess the efficacy of hydrologically isolated plots as the basis for direct in-field measurement of NO_3-N leaching from local soil types;
2. Evaluate drainflow recording/sampling techniques and equipment;
3. Collect some basic NO_3-N leaching data from a local soil type.

3.1 MATERIALS AND METHODS

3.1.1 SITE

The plot site was adjacent to the main buildings at Boghall Farm and comprised an area of rough, unfertilised grassland approximately 12x20 m with a slope of about 10%. The site was at an elevation of approximately 215 m (705 feet) with an easterly aspect. Further up the slope beyond the site was the farm car park, an arable field and then rough grazing pasture rising into the Pentland Hills to an elevation of 300 - 400 m (1 000 - 1 300 feet).

The soil on the site was an imperfectly drained sandy loam, classified as a Sourhope/Winton series intergrade. The parent material included soliflucted andesitic drift over glacial-till derived from Carboniferous sediment. Inspection of the soil profile (Table 3.1) during plot isolation had revealed that the subsoil, $C_{(g)}$ horizon, was well compacted with a measured bulk density of 1.77 g cm^{-3} (Dr A. Vinten, East of Scotland College of Agriculture, personal communication, 1986).

Table 3.1 : Typical soil profile description at Boghall Farm

Horizons (cm):

0 - 5	L	Undecomposed litter. Sharp change to:
5 - 40	A_h	Dark brown [7.5YR3/4]; Sandy loam; High organic matter and abundant roots. Gradual change to:
40 - 60	A_w	Dull reddish brown [5YR4/4]; Sandy loam; Less organic matter and fewer roots. Sharp change to:
60 - 100	B_w	Greyish brown [5YR4/2] with some gley patches [7.5YR6/1]; Sandy clay loam; Signs of root penetration and clay translocation. Sharp change to:
100 - 140	$C_{(g)}$	Dark brown [7.5YR3/4] with considerable gleying [5YR4/4]; Sandy clay loam; Platy; Signs of clay translocation. Gradual change to:
140+	C	Dark reddish brown; Loamy sand; Platy; Saturated; Significant quantities of gravel.

The history of the site was not known, but upon commencing plot isolation it had become evident that the site contained at least one old tile drain system at a depth of approximately 1.0 - 1.2 m.

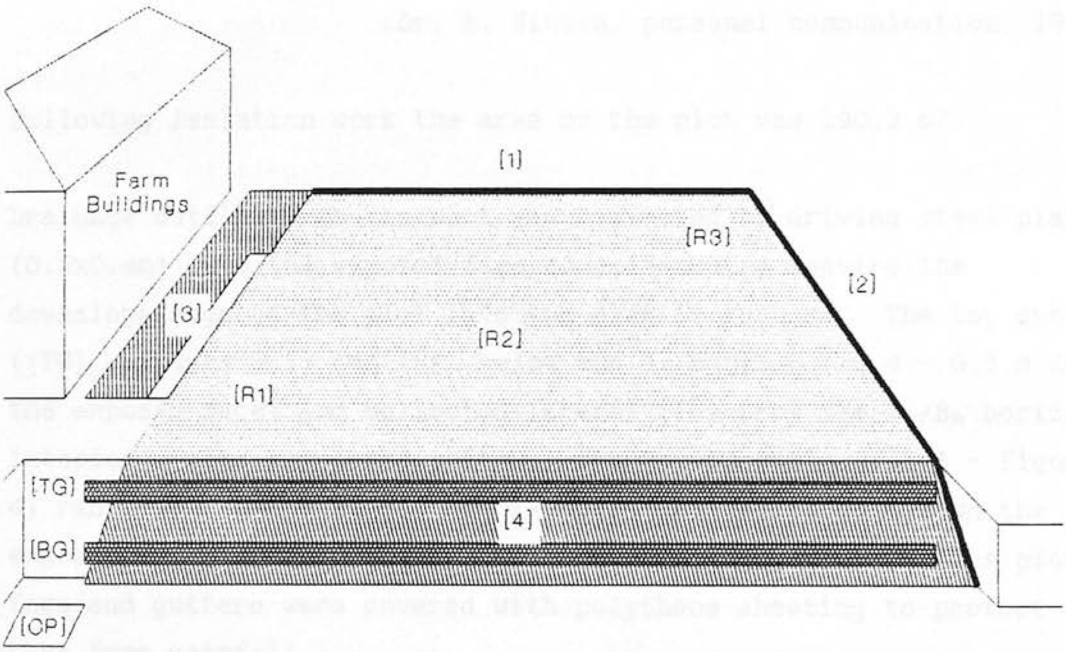
3.1.2 PLOT ISOLATION

Plot isolation had involved the following elements (refer to Figure 3.1):

[1] - installation of a 1.8 m deep isolation drain along the top edge of the plot, lined with continuous polythene sheeting on the downslope side (*i.e.* against the plot) and back-filled to the surface with gravel. The isolation drain discharged into an existing ditch;

[2] - the insertion of 'Enkadrain' (a proprietary drainage matting combining plastic sheeting and a permeable lining) to a depth of 1.0 m along one side of the plot;

Figure 3.1 : Stylised diagram of Boghall pilot plot showing the main elements of plot isolation and outflow collection (refer to text for annotation)



KEY TO DIAGRAM:

- Plot area

- Exposed plot face

- Guttering

[TG] – Top gutter
[BG] – Bottom gutter
[CP] – Concrete plinth
[R1] – Raingauges 1 – 3

Refer to text for numerical annotation

[3] - leaving an open ditch (approximately 1.0 m deep) between the plot and an adjacent barn;

[4] - cutting a clean face (approximately 1.5 m high) across the bottom edge of the plot and exposing the soil profile.

(Dr. A. Vinten, personal communication, 1986)

Following isolation work the area of the plot was 190.2 m^2.

Drainage outflow from the plot was collected by driving steel plates (0.3x0.4m) into the exposed face to guide water leaving the downslope edge of the plot into two plastic gutters. The top gutter ([TG] - Figure 3.1) ran just below the A_w horizon (≈0.4 - 0.5 m down the exposed face) and collected lateral flow from the A_w/B_w horizon interface, plus any surface flow. The bottom gutter ([BG] - Figure 4) ran below the level of the clay tile drains (≈1.25 m down the exposed face) and collected any deeper drainage outflow. The plot face and gutters were covered with polythene sheeting to protect them from rainfall.

Data was collected from the plot between 1 November, 1986 and 7 April, 1987 and ceased only because of problems experienced with the stability of the plot face. During a prolonged cold period (7-20 January, 1987) the face froze and became badly fractured, gradually collapsing over the next few months. It became apparent that significant amounts of water were being lost from the face due to the dislodgement of the steel plates and blockage of the guttering, and data collection was forced to stop.

3.1.3 PLOT TREATMENTS

The plot remained as a 'bare fallow' for the whole experimental period. It had been sprayed with the herbicide 'Roundup' (a.i.: glyphosate) in July 1986 and weed growth was minimal.

Two applications of 50g Br^- tracer (as $CaBr_2.H_2O$ in 20 litres of aqueous solution) were applied to the plot on 30 December, 1986 and 26 February, 1987.

3.1.4 FLOW MEASUREMENT

Outflow from the gutters was piped to two tipping bucket flow meters mounted on a concrete plinth adjacent to the plot ([CP] - Figure 3.1). Each flow meter (Field Drainage Experimental Unit, Ansty Hall, Trumpington, Cambs.) consisted of two steel 'buckets' mounted on a pivoting spindle supported in a rigid steel frame. The bucket capacity was set to approximately 5 litres (maximum flow rate of 1.1 litres s^{-1}).

The number of tips on each flow meter was counted by two magnetically actuated reed switches; one connected to a 12 volt portable chart recorder (each tip registered as a peak) and the other to a simple total run counter (providing back-up data in case of the chart recorder failing). Charts were collected every few days and outflow (mm) from the top and bottom gutters was recorded as 6 hourly totals.

The flow meter buckets were regularly calibrated in order to assist accurate calculation of plot drainage outflow. Ideally the calibration should have been dynamic (Calder and Kidd 1978), however, this was not practicable and may have been a source of error, particularly at high flow rates (discussed further in 3.3.2).

Rainfall was recorded as the mean of 3 syphonic raingauges ([R1], [R2] and [R3] - Figure 3.1) set diagonally across the plot. This was intended to overcome any problem of 'shadowing' from the adjacent farm buildings and also any calibration errors with individual raingauges.

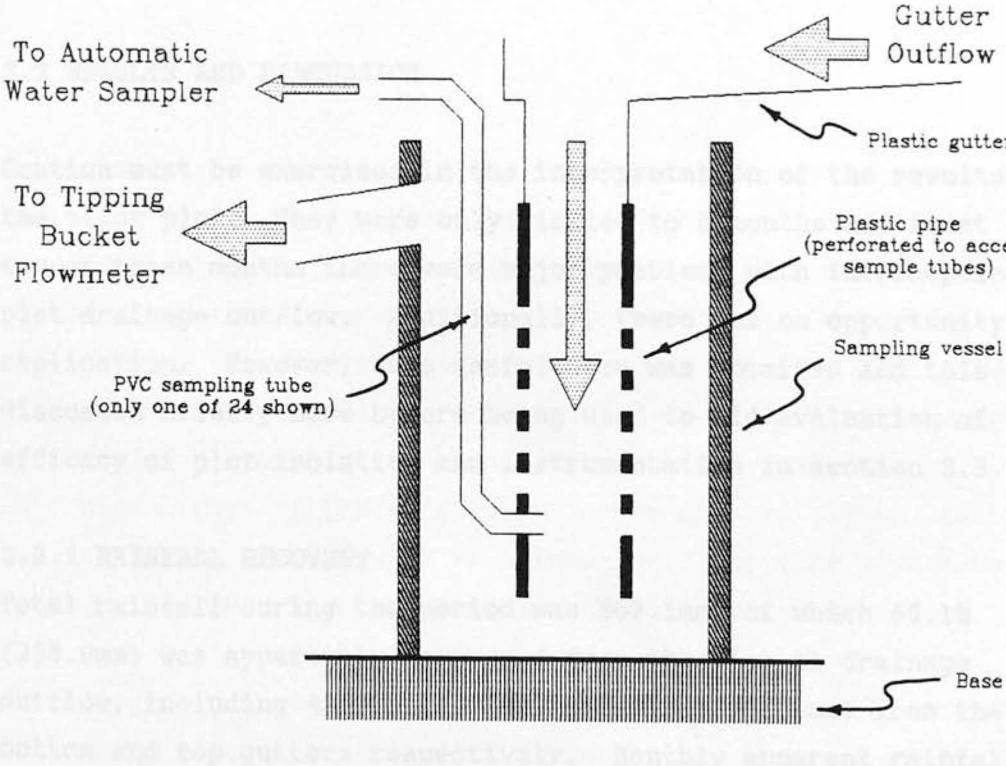
Towards the end of the experimental period a Cristie CD6 cassette data logger (Cristie Electronics Ltd, Stonehouse, Glos.) was introduced to the site for evaluation, together with a 0.25 mm tipping bucket raingauge (Environmental Measurements Ltd, Osney Mead, Oxford).

The data logger recorded hourly flow meter and raingauge tips on a magnetic tape cassette. This was collected weekly and the data transferred to floppy disk by interfacing a Cristie CS5 cassette terminal and BBC 'B' microcomputer. Software was specifically written to facilitate this transfer and also to collate the hourly data as weekly totals. The data was subsequently archived by transfer from the microcomputer to a VAX/VMS minicomputer.

3.1.5 WATER SAMPLING

Outflow from the gutters was sampled automatically using two specialist water samplers that sequentially released previously drawn vacuums in a series of 24 glass bottles (Automatic Liquid Samplers Ltd, Hanwell, London; SEC2 sampler - model 24). Each collection bottle was connected by a narrow bore PVC tube to a vessel through which the gutter outflow ran before discharging into the tipping bucket (Figure 3.2); when the bottle's vacuum was released a discrete water sample (≈ 500 ml) was collected.

Figure 3.2 : Schematic diagram of water sampling vessel



The water samplers included a proportional sampling facility (*i.e.* samples could be taken after a pre-determined number of bucket tips), but this proved extremely unreliable and was not used. Instead, sampling frequency was controlled by an electronic timer set at a fixed 5½ hour cycle (the maximum interval possible). It was intended that each 6 hour period used for recording outflow should include at least one water sample for the estimation of solute loading.

The water samples were regularly collected from the site and immediately analysed (or else frozen until analysis was possible). NO_3-N and NH_4-N levels (mg l^{-1}) were determined in duplicate sub-samples by continuous flow analysis, using the methods of Henricksen and Selmer-Olsen (1970) and Crooke and Simpson (1971) respectively. NH_4-N levels were insignificant and analysis was discontinued. Br^- levels were determined using an Orion specific-ion electrode.

The water sampler for the top gutter was not available until early December 1986, N losses from the top gutter prior to this were calculated using an approximated NO_3-N concentration based upon average NO_3-N levels in the top gutter outflow in mid-December.

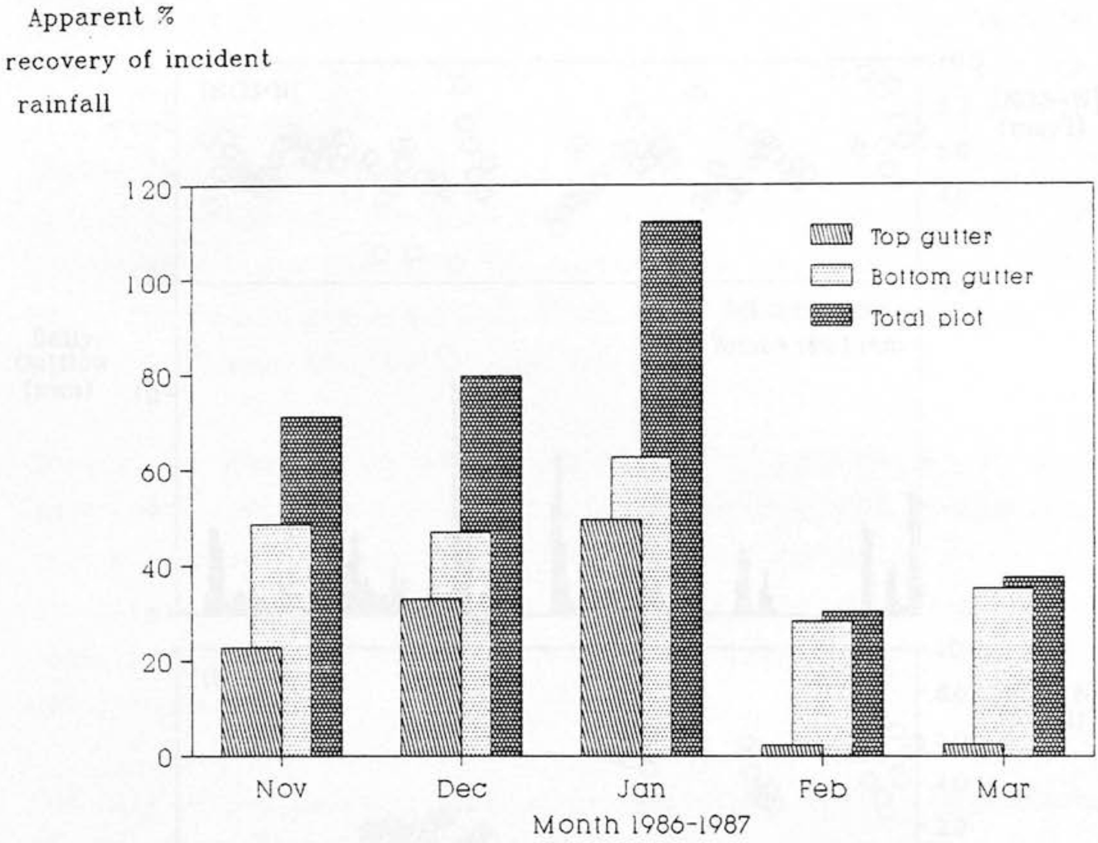
3.2 RESULTS AND DISCUSSION

Caution must be exercised in the interpretation of the results of the pilot plot. They were only limited to 5 months and in at least two of these months there were major problems with intercepting the plot drainage outflow. Additionally, there was no opportunity for replication. However, some useful data was obtained and this is discussed briefly here before being used to aid evaluation of the efficacy of plot isolation and instrumentation in section 3.3.

3.2.1 RAINFALL RECOVERY

Total rainfall during the period was 367.1mm, of which 65.1% (238.9mm) was apparently recovered from the plot as drainage outflow, including 43.7% (160.3mm) and 21.4% (78.6mm) from the bottom and top gutters respectively. Monthly apparent rainfall recovery was variable, ranging from 30.2 - 112.2% (Figure 3.3).

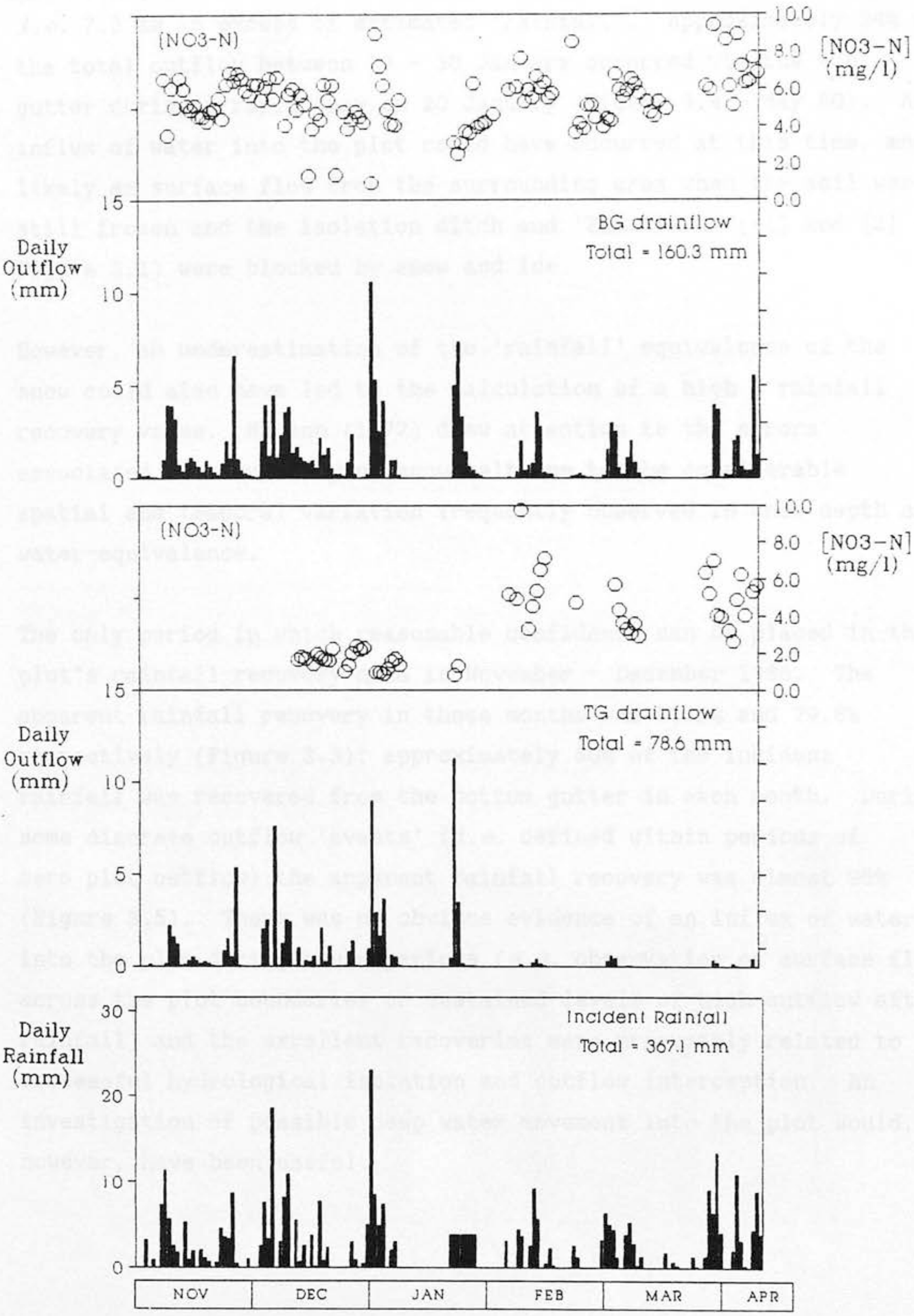
Figure 3.3 : Apparent % recovery of monthly incident rainfall from Boghall pilot plot



The lowest % recoveries coincided with the collapse of the exposed plot face during February and March 1987. The effect of the deteriorating plot face was most evident as a marked reduction in outflow from the top gutter (Figure 3.3 and 3.4) as the adjacent steel plates became dislodged and water ran down the plot face rather than into the top gutter. The efficacy of the bottom gutter was less affected by dislodgement of the steel plates and intercepted much of the outflow lost from the top gutter, but it rapidly became blocked with soil from the collapsing plot face and could not be kept clear for any significant period of time.

The occurrence of a relatively high % rainfall recovery in January (Figure 3.3) suggests an influx of water into the plot, but some caution must be exercised in drawing this as a conclusion.

Figure 3.4 : Daily rainfall (mm), outflow (mm) and mean daily NO_3-N concentration (mg l^{-1}) from Boghall pilot plot during the period 1 November 1986 to 7 April 1987.



A significant proportion of the precipitation in January was snow and its 'rainfall' equivalence, based upon snow depth on the plot, was estimated as 26.3mm. The snow began melting on 19 January and was gone by the 26 January. There was no further rainfall until 2 February. Total plot outflow between 19 - 30 January was 34.1 mm *i.e.* 7.8 mm in excess of estimated 'rainfall'. Approximately 34% of the total outflow between 19 - 30 January occurred via the top gutter during a rapid thaw on 20 January (Figure 3.4 - day 80). An influx of water into the plot could have occurred at this time, most likely as surface flow from the surrounding area when the soil was still frozen and the isolation ditch and 'Enkadrain' ([1] and [2] - Figure 3.1) were blocked by snow and ice.

However, an underestimation of the 'rainfall' equivalence of the snow could also have led to the calculation of a high % rainfall recovery value. Hudson (1977) drew attention to the errors associated with quantifying snow-melt due to the considerable spatial and temporal variation frequently observed in snow depth and water-equivalence.

The only period in which reasonable confidence can be placed in the plot's rainfall recovery data is November - December 1986. The apparent rainfall recovery in these months was 71.2% and 79.8% respectively (Figure 3.3); approximately 50% of the incident rainfall was recovered from the bottom gutter in each month. During some discrete outflow 'events' (*i.e.* defined within periods of zero plot outflow) the apparent rainfall recovery was almost 95% (Figure 3.5). There was no obvious evidence of an influx of water into the plot during these periods (*e.g.* observation of surface flow across the plot boundaries or sustained levels of high outflow after rainfall) and the excellent recoveries were presumably related to successful hydrological isolation and outflow interception. An investigation of possible deep water movement into the plot would, however, have been useful.

Figure 3.5 : Daily hydrograph for the Boghall pilot plot between 21 - 30 November 1986.

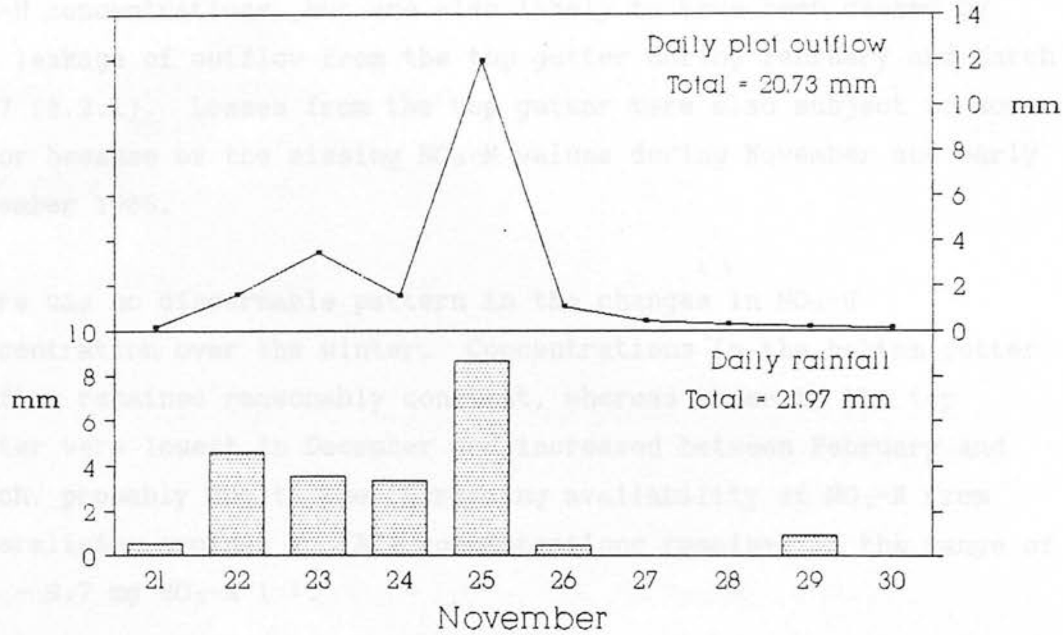
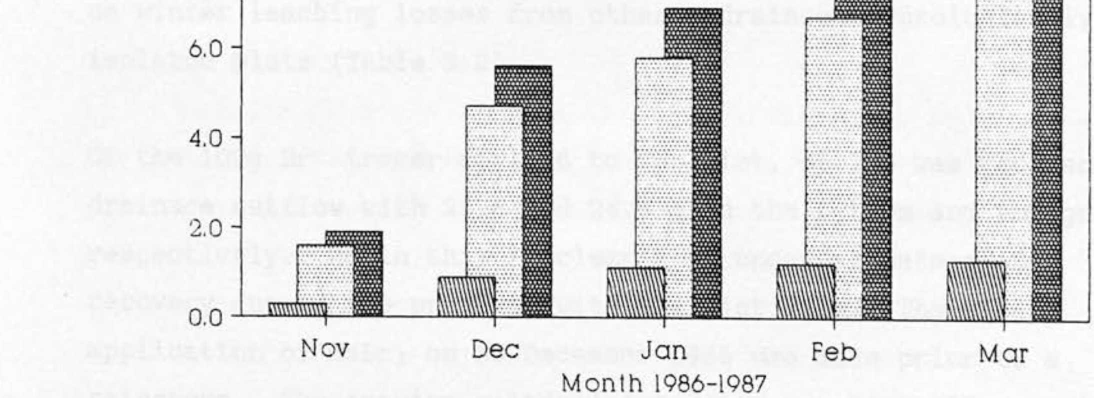


Figure 3.6 : Cumulative monthly NO_3-N leaching losses (kg ha^{-1}) from Boghall pilot plot.



3.2.2 SOLUTE LEACHING

Total NO_3-N leaching losses from the plot were equivalent to 10.3 kg N ha^{-1} and included 9.0 and 1.3 kg N ha^{-1} from the bottom and top gutters respectively (Figure 3.6). The higher losses from the bottom gutter were due to both greater outflow and generally higher NO_3-N concentrations, but are also likely to have been caused by the leakage of outflow from the top gutter during February and March 1987 (3.2.1). Losses from the top gutter were also subject to some error because of the missing NO_3-N values during November and early December 1986.

There was no discernable pattern in the changes in NO_3-N concentration over the winter. Concentrations in the bottom gutter outflow remained reasonably constant, whereas those in the top gutter were lowest in December and increased between February and March, probably due to the increasing availability of NO_3-N from mineralising organic N. All concentrations remained in the range of 0.8 - 9.7 mg NO_3-N l^{-1}.

The greatest single monthly loss was 3.7 kg N ha^{-1} in December and corresponded to the greater plot outflow during this month, notably that arising from the two biggest rainfall events of the winter (Figure 3.4). It is not possible to comment further on the pattern of NO_3-N loss because of the problems subsequently experienced with the plot face.

Total NO_3-N leaching losses measured from the plot were obviously an underestimate. However, they are broadly in line with reported work on winter leaching losses from other undrained, hydrologically isolated plots (Table 3.2).

Of the 100g Br^- tracer applied to the plot, 46.2 g was recovered in drainage outflow with 21.6 and 24.6 g in the bottom and top gutters respectively. Again this is clearly an underestimate of Br^- recovery due to the problems with the plot face. The first application of $CaBr_2$ on 30 December 1986 was made prior to a rainstorm. The ensuing rainfall intensity was high (13.4 mm in 6

Table 3.2 : Winter NO_3-N leaching losses from some individual undrained plots

Winter:	Treatment:	Plot discharge (mm):	NO_3-N loss (kg ha^{-1}):	Reference:
1979-80	Fallow after barley	224	17.0	Armstrong *et al.* (1983)
1978-79	Winter wheat	25	3.9	Harris *et al.* (1984)
1979-80	Winter wheat	45	11.8	*ibid.*
1986-87	Fallow	239	10.3	This work

hours) and 10.4 g of Br^- was immediately recovered from the top gutter, probably due mainly to surface run-off, with a further 5.3 g recovered from the top gutter during the following 12 hours.

Soil sampling to establish residual Br^- levels in the plot and the determination of 'unrecovered' Br^- at the end of the experimental period would have been useful in assessing the efficacy of the plot for the measurement of losses from a surface-applied solute. Nonetheless, accepting this limitation and the problems suffered with the plot face, it was still possible to establish that significant quantities of Br^- were recovered in plot outflow.

3.3 EVALUATION OF METHODOLOGY

3.3.1 PLOT ISOLATION

It is evident from the results that the best period for assessing the efficacy of plot hydrological isolation was November - December 1986. The mean apparent % rainfall recovery obtained during this period (76.7%) was encouraging and compared very favourably with the apparent rainfall recoveries reported for the same time of year from other undrained plots on heavy soil types (Table 3.3).

Table 3.3 : Apparent recovery of rainfall as drainage outflow from hydrologically isolated plots on heavy soil types

Period:	Rainfall (mm):	Plot discharge (mm):	Apparent rainfall recovery:	Reference:
Oct. 1979 - April 1980	467.4	224.4	48.0%	Armstrong *et al.* (1980)
Nov. - Dec. 1968	116.7	48.3	41.4%	Burke (1974)
Oct. - Nov. 1970	128.7	35.5	27.6%	*ibid.*
Jan. 1978 - Mar. 1979	175.0	25.0	14.3%	Harris *et al.* (1984)
Dec. 1979 - Mar. 1980	261.0	45.0	17.2%	*ibid.*
Nov. - Dec. 1986	183.9	141.1	76.7%	This work

Table 3.4 : Rainfall and plot outflow data for the Boghall pilot plot during November and December 1986

	November	December
Rainfall (mm)	66.0	117.9
Bottom gutter outflow (mm)	32.0	55.4
Top gutter outflow (mm)	15.0	38.7
Total outflow (mm)	47.0	94.1
'Unrecovered' rainfall:		
- total (mm)	19.0	23.8
- mean daily loss (mm)	0.63	0.77

The likely fate of the 'unrecovered' rainfall on the Boghall plot is uncertain. Assuming there was minimal loss at the plot face during November and December, incident rainfall could have been lost by evaporation or deep percolation. There is no evaporation loss data available for the site, but Snaebjornsson (1977) found that saturated conductivity (K_{sat}) of the 50 - 75 cm depth range of the Winton soil series (the pilot plot was a Winton/Sourhope intergrade soil) had a median value of 3 mm per day with a range of 1.3 - 7.6 mm per day. The mean daily losses of 'unrecovered' rainfall during November and December (Table 3.4) are well below these conductivity values and so could have occurred by deep percolation below the depth of plot isolation.

It would have been desirable to have conducted further work on the Boghall plot, including the estimation of potential evaporation losses and the measurement of hydraulic conductivity, in order to prepare a more complete water balance. This would have been useful, for example, in assessing the possibility that the plot had suffered some deep water influx during November and December.

Nonetheless, it seemed reasonable to suggest at the end of the experimental period that the use of the hydrologically isolated pilot plot had been successful in determining that: i) significant quantities of NO_3-N were lost from a fallow soil during the winter period; and ii) significant quantities of a surface applied solute were also lost. The principle of plot isolation was therefore considered appropriate for further leaching work on local soil types.

It was, however, apparent that:

a) the use of an exposed face was not appropriate (this was unlikely to be viable anyway at a normal field site);

b) plot isolation should continue to be made to a reasonable depth in order to intercept drainflow (including that from old drainage systems) and other water movement in the lower soil horizons. The higher recorded outflows and greater

NO_3-N losses from the bottom gutter on this plot suggest that it is not a reasonable assumption that the majority of water and solute movement occurs as lateral flow in these soil types.

3.3.2 INSTRUMENTATION

1. Drainflow measurement

Use of the tipping bucket flow meters was very successful. They proved to be robust and reliable, and no surcharging was noted. The maximum outflow recorded from a single gutter was 6.2 mm in 6 hours *i.e.* a mean flow rate of only 0.06 l s^{-1} which was well within their capacity, even assuming considerable temporal variation in flow rate.

Observation of the tipping buckets at Boghall identified a number of advantages which made their continued use in this project attractive (*e.g.* compared to V-notch weir systems):

1. they are simple in design, very easily maintained and require minimal expertise;
2. they give a simple digital output which is well suited to data logging devices and on-site integration (important for the immediate identification of problems with plot drainage);
3. they can be easily calibrated;
4. they can operate in a relatively shallow sump;
5. they can accurately measure low flow (Edwards *et al.* 1974).

According to other workers the main limitation of tipping bucket flow meters is reduced accuracy at high flow rates. This is due to a number of factors:

i) As flow rate increases an increasingly greater quantity of water is lost during the time taken for the bucket to move from one resting position to another. Calder and Kidd (1978) therefore suggested that tipping buckets should be calibrated dynamically over a reasonable range of flow rates, rather than statically;

ii) The inherent turbulence associated with high flow rates tends to increase the chance of premature tipping and also increases variation in the volume that initiates tipping (Edwards *et al.* 1974);

iii) Increasing quantities of water can be simply lost due to excessive splash at high flow rates, although the significance of this will depend greatly upon the height of the inlet pipe (Talman 1979).

Dynamic calibration was not practicable in this work and it was difficult to assess the potential errors that occurred as a result. Talman (1979) suggested that statically calibrated buckets are only accurate up to flow rates of 0.1 - 0.2 l s^{-1} (*i.e.* 1.9 - 3.8 mm hr^{-1} on a 190.2 m^2 plot). It is not clear whether these flow rates were reached at the Boghall pilot plot, but on larger plots at another site they could presumably be exceeded relatively easily.

One problem that was observed with the tipping buckets was freezing under low temperature. This not only included the freezing of water in the buckets, but also the freezing of empty buckets to their stops. Presumably this problem is no worse with tipping buckets than other flow measurement systems and was not significant in particularly cold weather since plot outflow had usually also ceased. However, on several occasions outflow measurements were lost as frosty weather froze the buckets, but not did not stop the plot outflow.

The use of a chart recorder was not appropriate for recording signals from the tipping buckets' reed switches. The recorder was not very robust and the peaks on the chart were often not clearly distinguishable, particularly at high flow rates when the peaks merged to form an illegible blur. The simple total run counters provided a useful back-up to the chart recorder, but were not compatible with the need to record against a time scale for the calculation of NO_3-N losses. The use of the data logger towards the end of the experimental period was very successful and the opportunity was taken for refinement of computerised plot data handling, analysis and archiving.

2. Water sampling

The automatic water sampling machines were reasonably effective and reliable at collecting spot samples on a fixed time interval from the plot outflows. The use of the proportional sampling facility on the automatic samplers would probably have contributed to a more reliable estimate of NO_3-N loading (*i.e.* by always sampling from a constant outflow volume), but would have required greater attendance at the plot site in order to check that the bottles had not all been used, particularly during periods of heavy rainfall. At least with the fixed sampling interval it was possible to plan a schedule for visiting the site. Ideally what was needed was a flow proportional sampler that was not limited by a fixed sampling capacity.

It was also possible that samples taken at very low flows may not have been truly representative of plot outflow, since it was unlikely that all of the residual water in the sampling vessel (5.0 - 7.5 l) was displaced between consecutive samples.

Overall, the samplers were unduly complex for the work being conducted on the plot. They required a time consuming procedure to load the bottles, evacuate them and subsequently collect the samples. This proved to be a problem in inclement weather since it necessitated opening the samplers and exposing the electronic timers, which subsequently proved sensitive to the wet and cold conditions.

Although other workers have apparently used automatic sampling machines very successfully (*e.g.* Harris *et al.* 1984), at the end of the experimental period it was obvious that a more straightforward sampling system was needed (*e.g.*. a simple flow-dividing device as used by Zwerman *et al.* 1972). It was felt that the automatic samplers should be reserved for those times when more intensive sampling was needed *e.g.* the preparation of detailed chemographs.

3.4 CONCLUSIONS

The results of the Boghall pilot plot suggested that hydrologically isolated plots could be used as the basis of direct in-field measurement of NO_3-N leaching losses from local soil types. The main points borne in mind when designing and establishing the main experimental site were:

1. Exposed plot faces should not be used;
2. Plot isolation should be deep enough to intercept all drainflow and potential deep water movement into plots;
3. Tipping bucket flow meters are a robust, reliable and accurate means of measuring plot outflow;
4. There is need for a simple flow proportional sampling technique;
5. Electronic data loggers can significantly ease data collection and subsequent analysis.

Despite problems with the plot face at Boghall, and consequently with drainflow collection, the measured NO_3-N winter leaching loss of 10.3 kg N ha^{-1} was well within the range of estimates from other research work of a similar nature.

CHAPTER 4 : GLENCORSE EXPERIMENTAL SITE - MATERIALS AND METHODS

4.1 SITE DESCRIPTION

A suitable field-site for the establishment of the hydrologically isolated plots was identified in March 1987 on part of the Edinburgh University's Bush Estate at Glencorse Mains Farm, near Penicuik, Midlothian (O.S. grid reference NT236627). The site was at an elevation of approximately 200 m (660 feet) with a south-easterly aspect. The soil was an imperfectly drained clay loam, classified as a surface water gley of Winton series (Ragg and Futty 1967), equivalent to a cambic stagnogley (after Avery 1980). The parent material was a mixed glacial till formed from Carboniferous sediments. This soil series occurs widely in South-east Scotland and is one of the most important arable soils in the Lothian region (Ragg and Futty 1967).

The site was surveyed and had a uniform slope of about 5%. Three 1.5 m deep pits were opened for soil profile descriptions (Plate 4.1) across the site and samples were taken for routine chemical and physical analysis. Details of the profile description and routine analysis are given in Tables 4.1 and 4.2.

Although soil variability was not investigated beyond observation and sampling from the three pits, some heterogeneity was evident across the site. Soil organic matter levels were particularly variable, especially in the plough layer, and this may have arisen from the sites' history as an experimental area (4.1.2). Overall, the soil organic matter levels were high in comparison with English soils (Stengel *et al.* 1984). Subsoil texture was also quite variable (sandy loam to silty clay loam), as is typical of many soils derived from glacial deposits.

It was decided that the site was suitable for hydrological isolation; namely that the subsoil was heavily compacted and water movement was predominantly as downslope interflow, with limited potential for deep percolation losses or rising artesian water.

Table 4.1: Representative soil profile description for the Winton soil series at East Flotterstone Field, Glencorse Mains Farm.

Horizons (cm):

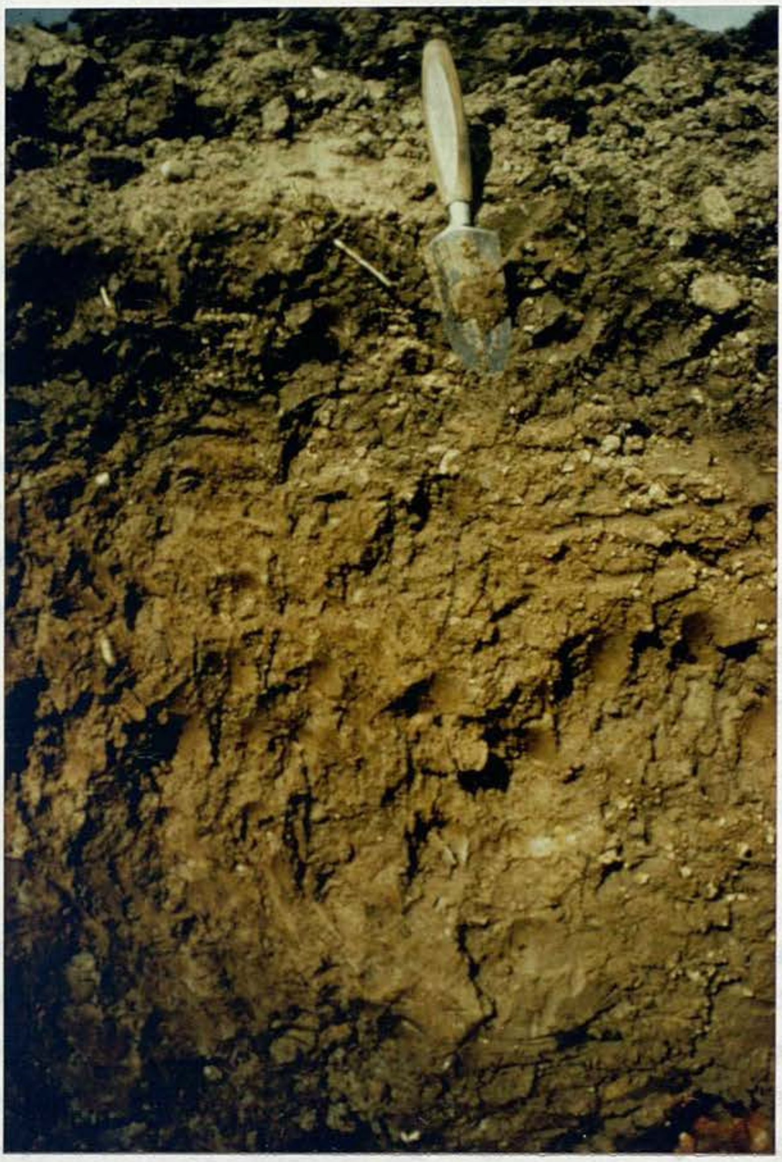
0 - 24	$Ap_{(g)}$	Dark greyish brown (10YR4/2) with few fine yellowish brown (10YR5/4) mottles; Clay loam; Moderate medium subangular blocky; Firm; Moist; Low organic matter; Common fine roots; Common small subangular stones; Sharp change to:
24 - 34	$B_{(g)}$	Brown (7.5YR5/2) with common medium strong brown (7.5YR5/6) mottles; Sandy silt loam; Moderate medium subangular blocky; Friable; Moist; No organic matter; Few fine roots; Common small subangular stones; Sharp change to:
34 - 65	$B2_{(g)}$	Reddish brown (5YR4/3) with common medium yellowish red (5YR4/8) mottles and common medium and coarse pinkish grey (5YR6/2) gley patches; Sandy clay loam; Firm; Moist; No organic matter; Very few fine roots; Common medium subangular stones, sandstones, many strongly weathered; Gradual change to:
65 - 110+	$BC_{(g)}$	Reddish brown (5YR4/3) with few medium yellowish red (5YR4/8) mottles and common fine and medium light grey (5YR7/1) gley patches; Sandy clay loam; Weak medium subangular blocky; Firm; Moist; No organic matter; No roots; Common medium and few large subangular stones, sandstones with few igneous rocks.

Acknowledgement: *Thanks to Frank Dry of the 'Macauley Land-Use Research Institute' for the profile descriptions.*

Table 4.2: Representative soil analyses for the Winton soil series at East Flotterstone Field, Glencorse Mains Farm.

Horizon	pH	Organic matter(%)	Mechanical analysis range (8 samples) Sand(%)	Silt(%)	Clay(%)	Bulk density at time of sampling (g cm^{-3})
$Ap_{(g)}$	6.2-6.7	4.8-6.0	31-53	25-44	21-25	1.19
$B_{(g)}$	6.3-6.7	0.6-1.9	14-57	21-52	17-33	1.47

Plate 4.1: Representative profile of the Winton soil series at East Flotterstone Field, Glencorse Mains Farm.



4.1.1 LAND USE CAPABILITY

The site was mapped as land use capability class 3.2 (average arable land producing good yields of a narrow range of crops) with a wetness limitation (Soil Survey of Scotland 1982). A more detailed classification of the actual site (using Bibby *et al*. 1982) was land use capability 4.1, suggesting the land was suited to rotational farming based on long ley grassland with short arable breaks (*e.g.* forage crops and cereals for livestock feed).

4.1.2 FIELD HISTORY

The field had been under spring barley for the previous 3 years, yielding 3.7 - 4.0 t grain ha^{-1} with fertiliser N inputs of 100 - 110 kg ha^{-1}. Prior to this it had been partly or wholly under potato trial work for many years. Although the land was unsuited to long arable rotations, spring barley was being regularly grown as a convenient cash crop whilst the field was being used for experimental tillage work by the Scottish Centre for Agricultural Engineering.

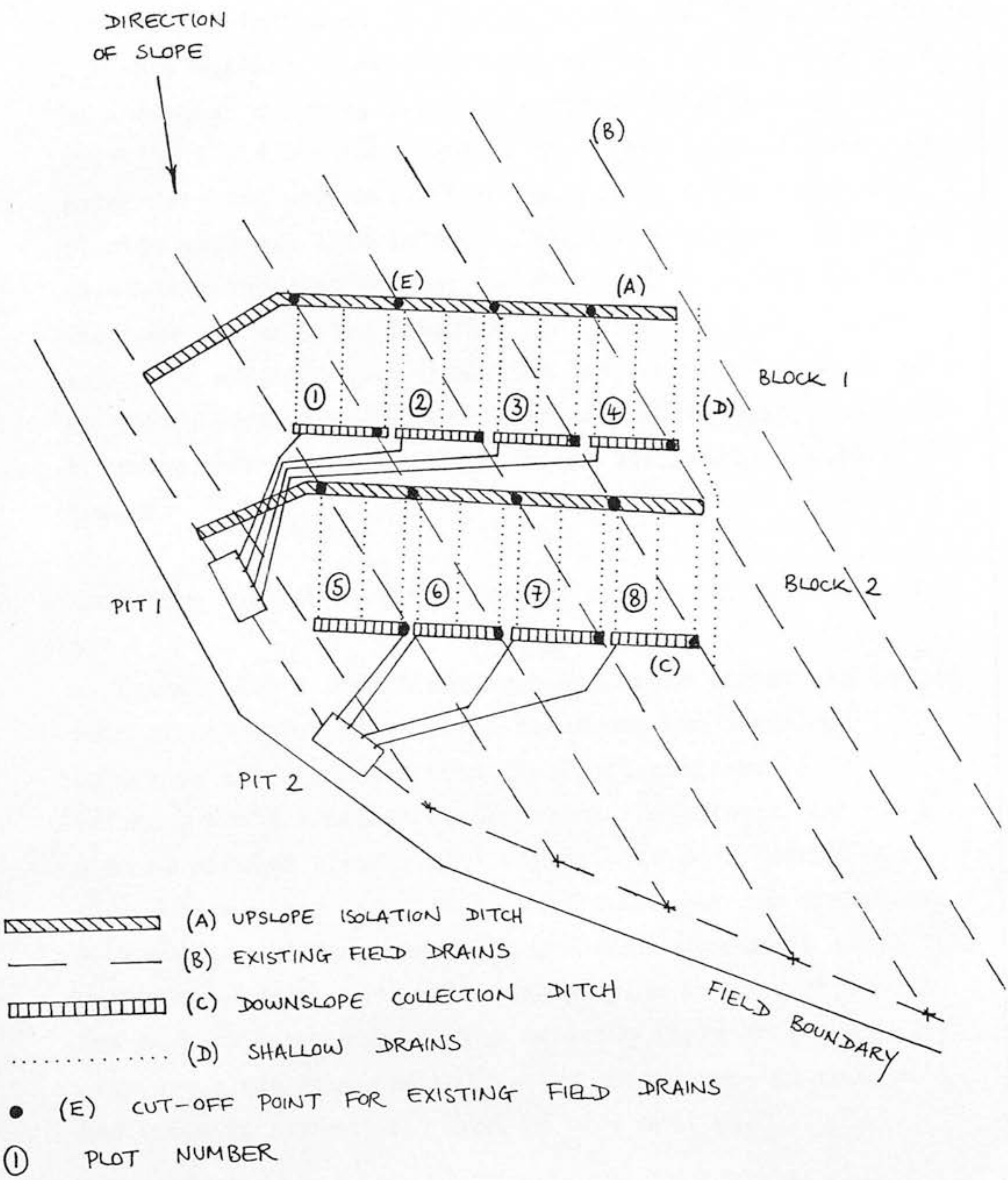
4.1.3 ANTECEDENT DRAINAGE SYSTEMS

A new drainage scheme had been installed in the field in September 1986, using 100 mm pipe, laid at 1 m deep and backfilled with gravel to the base of the plough layer. Drain spacing was 12 m. Plans of the scheme were available and it was incorporated as part of the plot isolation (4.2). During excavations for plot isolation a number of old clay tile drains were found ranging in depth between 0.5 - 1.0 m, suggesting the existence of at least 2 old drainage systems.

4.2 PLOT ISOLATION

Drainage work to establish eight 300 m^2 (15x20 m) hydrologically isolated plots began on March 30, 1987, and after delays due to heavy rain, was completed on April 25, 1987. During isolation work care was taken to avoid any vehicle traffic across the plot areas. The majority of the work was done by a local drainage contractor under the guidance and supervision of Dr A. Vinten and myself. It should however be noted that the complex nature of the drainage work was quite new to the contractor and mistakes were apparently made, particularly when not under direct supervision. This is not intended as criticism of the contractor, but rather a statement of one of the problems associated with such work. Interestingly, similar problems were noted almost 70 years ago by Hendrick (1921) in an account of the construction of the Craibstone lysimeters near Aberdeen.

Figure 4.1: The overall scheme of hydrological plot isolation at East Flotterstone Field, Glencorse Mains



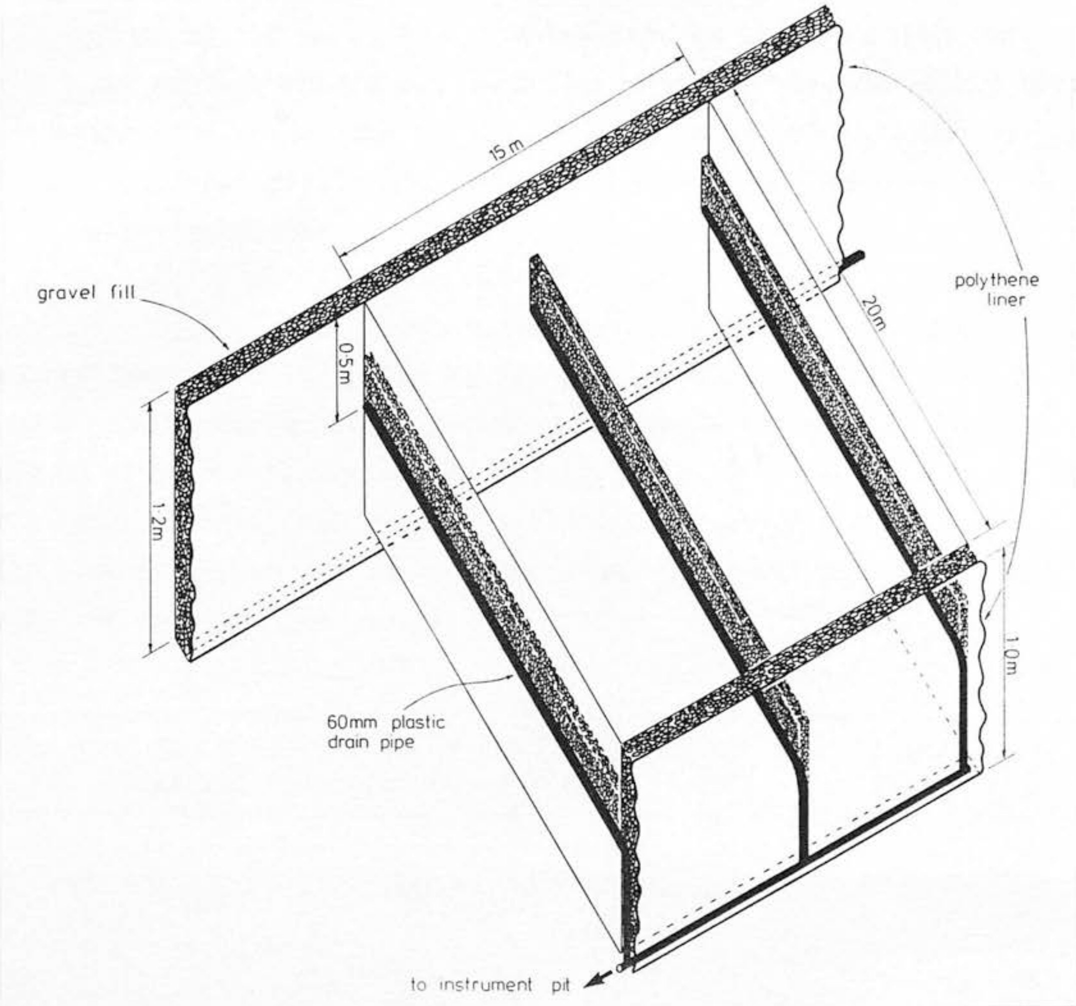
Plots were arranged across the sloping site in 2 blocks of 4 and configured in such a way that the recently installed field drains ran diagonally across each plot (Figure 4.1). Hydrological isolation for each block of plots consisted of 4 elements (summarised in Figures 4.1 and 4.2) :

1) An **upslope isolation ditch** running across the top of each block
   - a trench was excavated across the top of each block of plots to a nominal depth of 1.2 m in order to intercept the existing field drainage systems and prevent the movement of water into the plot area from upslope. A 60 mm slotted plastic pipe was laid in each trench and drainage water was piped to a suitable outfall in the recently installed field drainage system. The trenches were lined with continuous polythene sheeting (500 gauge) on the downslope side in order to prevent drainage of water into/out of the plot area. The trenches were then backfilled to the soil surface with gravel.

2) A **downslope collection ditch** running across the bottom of each plot
   - a separate 1.0 m deep trench was excavated across the bottom edge of each plot in order to intercept the downslope interflow and drainflow from the short sections of the existing field drainage systems that ran through the plots. A 60 mm slotted plastic pipe was laid in each trench, and plot drainage was piped to an instrument pit for drainflow measurement and sampling. The trenches were again lined with polythene sheeting on the downslope side and backfilled to the surface with gravel. The existing field drains running away downslope from the collection drains were plugged with bentonite to prevent any loss of plot drainage.

3) Three **shallow drains** running perpendicular to the collection drain in each plot
   - three 50 cm deep slit trenches were dug perpendicular to the collection trench in each plot using a tractor-mounted powered rotary slit trencher. Two of the trenches were

Figure 4.2: The main elements of hydrological isolation for a single plot



situated at the sides of each plot, and the third ran up the middle of the plot to give a spacing of approximately 7.5 m. A 60 mm slotted plastic pipe was laid in the slit trenches and connected to the slotted pipe in the collection ditch. The slit trenches were then back-filled to the surface with gravel. These shallow drains were intended to isolate the plots from each other by preventing any surface runoff and interflow across the slope, as well as generally improving plot drainage. Since the plot edges ran almost exactly perpendicular to the slope contours, and given the imperfectly drained nature of the soil, it was thought unnecessary to install deeper drains, which would have caused greater disruption to the plot areas.

4) **Shallow isolation drains** along the edges of the whole experimental area
   - shallow drains similar to 3) above were installed along the edges of the whole experimental area in order to intercept any surface runoff and interflow moving across the slope into the plot areas. The shallow drains discharged into the gravel-backfill of the existing drainage system outside the experimental area.

Once the plot area was ploughed, the effectiveness of the gravel backfill installed in 3) and 4) was partially lost, and it became evident in the Autumn 1987 that there was some transfer of water between plots due to surface runoff (5.3.1). Therefore, in November 1987 vehicle wheel ruts were made along the line of the shallow drains at the edges of the plots in order to divert surface runoff downslope into the plot collection drains.

Plate 4.2: The final stages of plot isolation at East Flotterstone Field, Glencorse Mains Farm

4.3 EXPERIMENTAL DESIGN

An immediate limitation of the work as planned (1.5) was the relatively short time available to investigate the N dynamics of a leguminous N input at the experimental site. It was therefore felt that the most appropriate action was to establish a leguminous green manure crop as soon as possible for incorporation in late Summer 1987.

Two forage legumes were identified for use as short-term green manures (Dr. J. Frame, West of Scotland College of Agriculture, personal communication, 1987):

- Red Clover (*Trifolium Pratense* L.);
- Forage Peas (*Pisum arvense* L.).

Of these 2 legumes, red clover is most commonly used as a green manure in the UK (Wyartt 1982). However under local conditions, forage peas are a more reliable crop (Mr. M. Morrison, East of Scotland College of Agriculture, personal communication, 1987) and were thus chosen as the green manure for this work.

Winter barley was established in Autumn 1987 as the main test crop. A winter cereal was chosen in order to ensure maximum utilisation of the legume-derived N (Mann 1958). With the introduction of better varieties, winter barley is an increasingly important crop in Scotland, notably because of its relatively high yield potential and early harvest compared to spring barley (SAC 1985).

The N fertiliser rates applied in this work were based upon SAC (1987a) recommendations for soils of low N status. The reduced N rate was an arbitrary 50% of the recommended N rate.

Winter forage rye (*Secale cereale*) was chosen as the over-wintering cover crop because of its proven efficacy as a rapidly growing catch crop under local conditions (Morrison *et al.* 1988). It is also now an approved species for use in Nitrate Sensitive Areas in England and Wales requiring winter cover up until 1

February (MAFF 1990c). An over-wintering legume such as trefoil (*Medicago lupulina*) was considered as the cover crop, however there was little experience of the use of such legumes under local climatic conditions (Mr. M. Morrison, personal communication, 1987) and it was suggested that winter hardiness might be limited. Problems with poor growth have also been reported when over-wintering trefoil on heavy soils (Barney 1987).

4.3.1 EXPERIMENTAL TREATMENTS AND CROPPING SEQUENCE

Soon after the plot isolation work was completed in April 1987, 4 treatments were established on the plots. Each treatment was replicated twice in a 'randomised block design'. The treatments were :

1. **spring barley + recommended N application**
   (120 kg N ha^{-1} - as seedbed dressing);
2. **spring barley + reduced N application**
   (60 kg N ha^{-1} - as seedbed dressing);
3. **spring barley + zero N application;**
4. **forage peas + 'starter' N application**
   (20 kg N ha^{-1} - at full emergence).

The N treatments were established on the spring barley primarily for the sake of obtaining some preliminary yield/N uptake for the site, including the investigation of soil- and fertiliser-N dynamics using ^{15}N-labelled fertiliser.

The instrumentation for drainflow measurement and sampling was still being installed during the summer of 1987 and so no comprehensive leaching data was collected.

The main experimental period was from September 1987 to April 1989 and included 2 specific periods. Firstly, the growth and harvest of the winter barley test crop, and secondly the 'winter fallow' period between harvest of the winter barley and establishment of the subsequent spring cereal. The following treatments were established on the winter barley :

1. **recommended N application**

   (150 kg N ha^{-1} - as split top dressings in Spring 1988);

2. **reduced N application**

   (75 kg N ha^{-1} - as above);

3. **zero N application;**

4. **legume N - derived from the incorporated forage peas.**

The over-wintering cover crop followed the winter barley receiving the reduced N application so that comparison could be made with the 'winter fallow' following both the zero and recommended N applications.

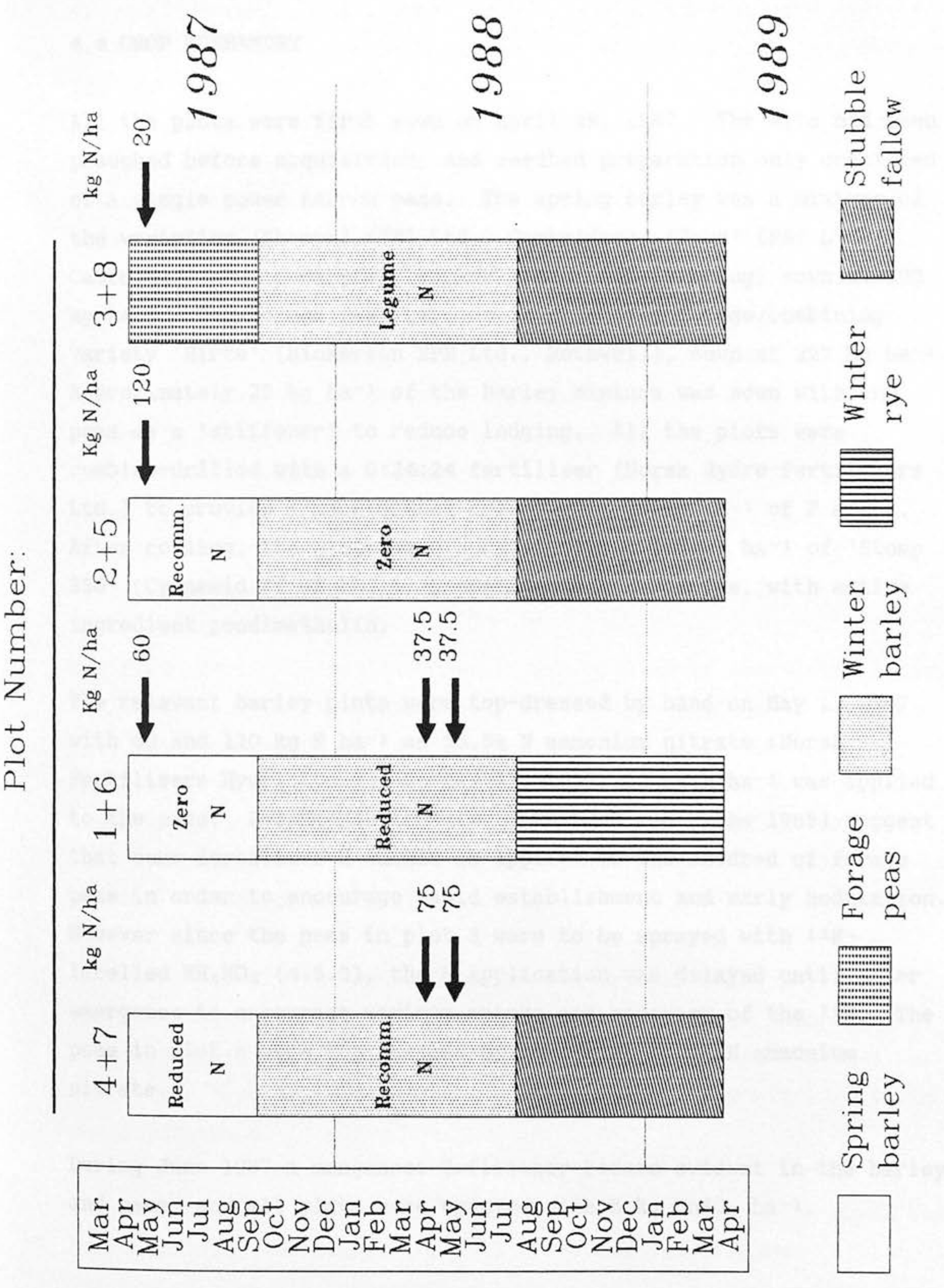
The sequence of crops and experimental treatments within the individual plots is summarised in Figure 4.3 (full husbandry details are given in section 4.4).

4.3.2 THE ALLOCATION OF 'CONTROL' PLOTS

Between April and September 1987 Plots 1 and 6 were the control zero N plots in the spring barley. In March 1988 when the fertiliser N was applied to the winter barley, the zero N treatments were re-allocated to Plots 2 and 5 which had previously received the recommended N rate on the spring barley. This re-organisation was done largely to compensate for the problems that were becoming evident with the plot isolation (4.5.4) by re-distributing the fertiliser N treatments across the troublesome plots.

Potentially, this complicated the use of a zero N control for the whole of the main experimental period from September 1987 to April 1989. It was considered leaving Plots 1 and 6 as the controls between September 1987 and March 1988, and then switching the controls to Plots 2 and 5 at the time of fertiliser application. However, such a switch would have introduced a further element of spatial variability problem into any calculations involving the zero N control, particularly in N balance calculations where the effect of spatial variability is additive during combination of soil mineral N, crop N and NO_3-N leaching measurements.

Figure 4.3: Summary of cropping and experimental treatments at the Glencorse experimental site, April 1987 - April 1989



Plots 2 and 5 were therefore used as the control plots throughout the main experimental period and the possible residual effects of fertiliser application to the preceding spring barley crop were ignored.

4.4 CROP HUSBANDRY

All the plots were first sown on April 29, 1987. The site had been ploughed before acquisition, and seedbed preparation only consisted of a single power harrow pass. The spring barley was a mixture of the varieties 'Sherpa' (PBI Ltd., Cambridge), 'Tyne' (PBI Ltd., Cambridge) and 'Camargue' (Booker Seeds Ltd., Feering) sown at 200 kg ha^{-1}. The forage peas were an intermediate forage/combining variety 'Birte' (Nickerson RPB Ltd., Rothwell), sown at 227 kg ha^{-1}. Approximately 25 kg ha^{-1} of the barley mixture was sown with the peas as a 'stiffener' to reduce lodging. All the plots were combine-drilled with a 0:24:24 fertiliser (Norsk Hydro Fertilisers Ltd.) to provide a maintenance dressing of 39 kg ha^{-1} of P and K. After rolling, the plots were sprayed with 4 litres ha^{-1} of 'Stomp 330' (Cyanamid of GB Ltd.), a pre-emergent herbicide, with active ingredient pendimethalin.

The relevant barley plots were top-dressed by hand on May 1, 1987 with 60 and 120 kg N ha^{-1} as 33.5% N ammonium nitrate (Norsk Fertilisers Hydro Ltd.). On May 22, 1987, 20 kg N ha^{-1} was applied to the peas. Recommendations (*e.g.* Whytock and Frame 1985) suggest that some fertiliser N should be applied to the seedbed of forage peas in order to encourage rapid establishment and early nodulation. However since the peas in plot 3 were to be sprayed with ^{15}N-labelled NH_4NO_3 (4.5.3), the N application was delayed until after emergence to encourage maximum uptake and recovery of the ^{15}N. The peas in plot 8 were top-dressed by hand with 33.5% N ammonium nitrate.

During June 1987 a manganese deficiency became evident in the barley and peas, and all plots were sprayed with 5 kg $MnSO_4$ ha^{-1}.

The forage peas were incorporated into the soil according to established guidelines (Schmid and Kläy 1982). The peas were chopped with a tractor-mounted rotary-mower on August 19, 1987 at the full bloom/flat pod stage after 16 weeks growth. The chopped material was left on the soil surface as a mulch until September 4, 1987, then incorporated into the soil using a rotary-cultivator. The depth of incorporation was only 10 - 15 cm in order to encourage an initial aerobic decomposition.

The spring barley was cut at maturity on September 17, 1987 and burnt *in situ* in the plot areas.

All the plots were conventionally ploughed on September 22, 1987 and after seedbed preparation (2 passes with a power-harrow) were sown with the winter barley variety 'Marinka' (Nickerson RPB Ltd., Rothwell) on September 28, 1987, at a seed rate of 210 kg ha^{-1}. The barley was combine-drilled with a 0:19:19 fertiliser (SAI Ltd., Edinburgh) to provide a maintenance dressing of 60 kg ha^{-1} of P and K. After rolling the plots were again sprayed with 6 litres ha^{-1} of the pre-emergent herbicide 'Stomp 330' (Cyanamid of GB Ltd.).

The first fertiliser split-dressing was applied to the winter barley on 29 March, 1988 at approximately GS 23 (the crop was starting to show the first signs of spring growth). Fertiliser was spread by hand as 34.5% N ammonium nitrate (ICI Fertilisers Ltd.) to plots 1, 4 and 6. Plot 7 was sprayed with an aqueous solution of ^{15}N-labelled NH_4NO_3 in order to look at the leaching of fertiliser-derived N from the whole plot (4.5.3).

The second fertiliser split-dressing was applied on 19 May, 1987 at approximately GS 43 (booting). This was rather later than recommended (*i.e.* GS 30-31 - beginning of stem extension). All fertiliser treatment plots received 34.5% N ammonium nitrate at the second dressing.

The winter barley was cut at maturity on July 26, 1988 and again was burnt *in situ*. On August 10, 1988 a seedbed was prepared in plots 1 and 6 with a single power-harrow pass. The winter forage rye

Plate 4.3: Experimental treatments established on the winter barley at East Flotterstone Field in May 1988

variety Rheidol (PBI Ltd., Cambridge) was broadcast by hand onto the seedbed at a rate of approximately 250 kg ha^{-1}, then harrowed in and rolled. All other plots were left as a stubble fallow.

All the plots were conventionally ploughed on March 17, 1989 and the experimental period finished on April 20, 1989 when spring barley was established in the plots and a new experimental programme started (details of further work at the site are contained in Vinten *et al.* 1991).

4.5 PLOT HYDROLOGY AND THE MEASUREMENT OF NITRATE-N LEACHING LOSSES

4.5.1 DRAINFLOW AND RAINFALL MEASUREMENT

The drainflow from each plot collection ditch (4.2) was piped to 2 instrument pits (one for each block), in which tipping bucket flow meters (3.1.4) were installed in the Summer 1987.

The flow meter 'buckets' were regularly calibrated throughout the experimental period in order to calculate weekly drainflow from the number of tips. The 'bucket' calibrations did vary slightly, primarily due to the flow meters working loose on their fixed mountings. The mean variation between consecutive bucket calibrations was, however, only in the region of 2%, so changes in 'bucket' volume were unlikely to be a major source of error in drainflow measurement. As noted at the Boghall Farm Pilot Plot (3.3.2) the flow meters should ideally have been dynamically calibrated, however this was again not practicable.

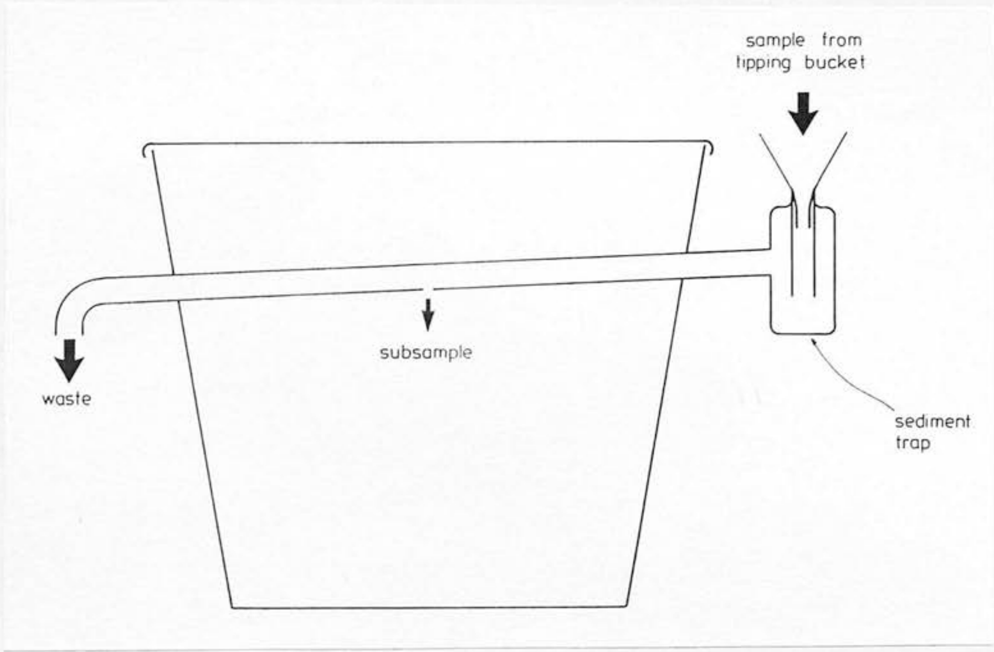
Rainfall was measured using two tipping bucket raingauges (3.1.4) mounted on concrete plinths at either end of the central discard area between the plot blocks (Figure 4.1). The raingauges were dynamically calibrated in the laboratory before installation. Potential evaporation data for the site was supplied by the Meteorological Office.

Data from the tipping bucket flow meters and raingauges were recorded on an hourly basis on Cristie CD6 multi-channel cassette data loggers (one data logger per block of plots). Cassettes were collected from the data loggers every 2 - 3 weeks, and the data was transferred from cassette to floppy disk (3.1.4). Weekly rainfall and plot drainflow measurements were collated and stored using an ORACLE relational database system running on a VAX/VMS minicomputer.

The Cristie data loggers became problematic during September/October 1988 and were replaced by electronic total run counters (Field Drainage Experimental Unit, Ansty Hall, Trumpington, Cambs.) from which data was recorded weekly.

The magnetically actuated reed switch on the tipping bucket flow meter for Plot 5 occasionally gave multiple counts for single bucket tips. The occurrence of multiple counting was identified from periods of low flow (*i.e.* when the data logger recorded an infeasible pattern of two or more tips per hour alternating with zero tips). A correction factor (≈ 0.5-0.6) was accordingly calculated and applied to the plot drainflow before any other correction procedures were used (4.5.4 and 4.5.5).

Figure 4.4: Cross-sectional diagram of the simple flow-dividing device used for sampling plot outflow



4.5.2 DRAINFLOW SAMPLING

Drainflow spot samples were collected manually throughout the Summer 1987. Routine drainflow measurement and sampling did not begin until September 4, 1987 with the incorporation of the forage peas in the legume N treatment, and final installation of all the field instruments.

The flow weighted mean NO_3-N concentration of the individual plot drainage was measured using a simple flow-dividing device (Figure 4.4 and Plate 4.4). This consisted of a closed plastic bucket with a plastic pipe running through it which carried a sample of the water discharged from one side of the tipping bucket flow meter. A small hole in the pipe allowed some of the sample (approximately 10 ml) to flow into the collecting bucket. Aliquots of the plot drainflow therefore accumulated in the collecting bucket, and a single sample was collected from the bucket at the end of each sampling period (approximately every week, but more frequently during periods of high drainflow). The volume of each aliquot was

Plate 4.4: Tipping bucket flow meter and simple flow-dividing device for sampling plot drainflow (seen installed in instrument pit)

likely to be variable, however this was considered to be unimportant because the number of aliquots collected was so large (≈300 for 10 mm drainflow).

NO_3-N and NH_4-N levels in the water samples were determined by continuous flow analysis (3.1.5). NH_4-N levels were insignificant and analysis was discontinued in December 1987. By careful recording of the time and date at which samples were collected, NO_3-N concentrations were precisely matched to drainflow volumes and plot NO_3-N loadings were accurately calculated. The ORACLE database (4.5.1) was used for all routine calculations and storage/retrieval of results.

The main advantage of this sampling system, compared to the automatic spot samplers used at the Boghall Pilot Plot (3.1.5), was that only one sample was needed for analysis to obtain an unbiased estimate of the flow-weighted average NO_3-N concentration during the sampling period. The sampling system was also extremely cheap and very easy to operate and maintain.

One potential problem with the system was the effective inhibition of biological activity in the buckets. Harris *et al.* (1984) reported that a final concentration of 1% phenol in a collected sample was sufficient to minimise biological activity before analysis. In this work, however, it was not possible to predict the final concentration of biocide in a sample because the final sample volume was always unknown. Instead 5 ml of chloroform was routinely added to the buckets at the start of each sampling period, the principle being to limit the activity of any micro- or macro-organisms already present in the buckets (Allen *et al.* 1974). The buckets were also regularly cleaned. The efficacy of this approach was not investigated in detail, but there was no evidence of extensive biological activity (*e.g.* algal growth) in the buckets or interference with N analysis.

4.5.3 ^{15}N TRACER EXPERIMENTS

Two ^{15}N tracer experiments were conducted to directly measure the contribution of legume- and fertiliser-derived N to NO_3-N leaching losses:

(i) On May 22, 1987 the 'starter' N application (4.4) to the forage peas in Plot 3 was applied as an aqueous solution of double-labelled 2.0 atom% $^{15}NH_4{}^{15}NO_3$ at the rate of 20 kg N ha^{-1}. A total of 20 l solution was sprayed onto the emergent crop using a knapsack sprayer. The objective was to label the forage peas sufficiently for investigation of the contribution of legume-derived N to the NO_3-N leaching losses from Plot 3;

(ii) On March 29, 1988 the first split of the fertiliser N application to the winter barley in Plot 7 (recommended N treatment) was applied as an aqueous solution of double-labelled 0.7 atom% $^{15}NH_4{}^{15}NO_3$ at the rate of 75 kg N ha^{-1}. A total of 20 l solution was again applied to the plot using a knapsack sprayer.

All ^{15}N-labelled fertiliser solutions were prepared by mixing double-labelled 5.0 atom% $^{15}NH_4{}^{15}NO_3$ with calculated quantities of NH_4NO_3 of normal isotopic composition (0.3663 atom%). Sub-samples of the fertiliser solutions were taken for isotopic analysis to confirm the actual ^{15}N enrichment achieved; results were satisfactory on all occasions.

After analysis for NO_3-N concentration (3.1.5) drainflow samples (≈500 ml) were reduced in volume by evaporation on a sand bath and steam distilled in the presence of both MgO and Devarda's alloy to reduce NO_3^- to NH_4^+. The collected distillate was acidified (1M sulphuric acid) and then evaporated to dryness on a sand bath before being analysed for $^{15}NO_3$ enrichment using a VG Isogas MM622 mass spectrometer linked to a Carlo-Erba 1400 automatic N analyser. $^{15}NH_4$ enrichment was not determined separately since concentrations were very low. All samples were analysed in duplicate.

The proportion of NO_3-N in the drainage water derived from ^{15}N-labelled fertiliser ($\%N_{dff}$) was calculated by the following equation (Bergström 1987):

$$\%N_{dff} = \frac{\text{atom\% } ^{15}\text{N excess (water sample)}}{\text{atom\% } ^{15}\text{N excess (fertiliser)}} \times 100 \qquad (10)$$

The atom% ^{15}N excess was calculated as follows:

$$\text{Atom\% } ^{15}\text{N excess} = \text{atom\% } ^{15}\text{N (sample)} - \text{atom\% } ^{15}\text{N (control)} \qquad (11)$$

where the control value was atmospheric ^{15}N enrichment (0.3663 atom%).

Atmospheric enrichment was used to calculate atom% excess for all ^{15}N experimental work conducted in this project *i.e.* the direct measurement of fertiliser N leaching, N_2 fixation (4.6.2), crop fertiliser N uptake (4.6.3) and residual fertiliser N (4.6.4).

4.5.4 DRAINFLOW RECOVERY PROBLEMS AND REMEDIAL ACTION

It soon became obvious after the beginning of the experimental period that the apparent recovery of incident rainfall from the 8 plots was very variable (5.3.1). Three distinct problems became evident from the recorded data and field observations:

a) **a rapid inflow of water into some plots during heavy rainfall**
   - indicated by short periods when the apparent recovery of the incident rainfall exceeded 100% (Plot 1 and possibly Plot 5). This problem was mainly attributed to surface run-off which was observed on several occasions to transfer water between plots and from the surrounding area into the plot area;

b) **a sustained inflow of water into some plots**
   - indicated by long periods of drainflow when the apparent recovery of incident rainfall greatly exceeded 100%. This problem was particularly evident in Plot 6 as an elevated baseflow during periods of low rainfall;

c) **a loss of drainage water from the collection system of some plots**
   - indicated by long periods of low rainfall recovery, with measurable drainflow only occurring after relatively large rainfall events (most notable in Plot 3 and to a lesser extent in Plots 4 and 8).

The following remedial action was therefore taken:

1) *March 1988*
   - surrounding area was cultivated to increase surface storage capacity and reduce surface run-off into plot area (it was subsequently sown to grass);
   - re-excavation (from outside the plot area) of all unions between downslope plot collection ditches and the existing field drainage system, in order to ensure that the field drains running away from the plots had been effectively cut and plugged. Unfortunately, the field drains running out of Plots 2, 3 and 7 still appeared to be carrying water. They were cut and plugged with bentonite;

2) *April 1988*
   - the drainage recovery from Plot 3 was still not improved by the

March 1988 work and so the union of the plot collection ditch and existing field drainage system was again re-excavated (this time from within the plot area). This revealed that the field drain leaving the plot actually ran beneath the collection ditch and was not connected to it. The field drain was therefore cut again and connected to the gravel backfill of the plot collection ditch. It was seen to be carrying water at this time and was apparently discharging into the collection ditch. The remaining section of field drain leaving the plot was plugged with bentonite;

3) *September 1988*

- Plot 3 was still problematic at this time, and so a 1.5 m trench was dug slightly downslope and parallel to the plot collection ditch in an attempt to identify any leakage from the ditch. An old clay tile drain at about 1.0 m depth was still active and was plugged with bentonite. No other potential leakage was observed;

- a similar trench was dug within Plot 6, downslope of the isolation ditch, in order to try and identify any movement of water into Plot 6. The existing field drain in the top of Plot 6 was still active suggesting that it was transferring water into the plot from the isolation ditch. The existing field drain was cut and plugged;

- when excavated in March 1988, the existing field drain leaving Plot 8 did not appear to be carrying any water and no remedial action was taken. However, because drainage from this plot was still relatively low, the existing field drain was again excavated (outside the plot area) and cut and plugged;

4) *November 1988*

- installation of 20 cm deep slit trenches along the lines of the shallow drains seperating the plots, in order to replace the vehicle wheel ruts and prevent transfer of water between plots by surface run-off;

- installation of a 40 cm deep permeable barrier (Enkadrain drainage matting) to the soil surface, along the boundary between plots 1 and 5 and the surrounding discard area. This was intended to intercept and divert the movement of extraneous surface water away from the plots.

The problems with drainflow recovery from Plot 3 were not resolved until the autumn 1989 when the main experimental period had finished. It became apparent that the pipe from the collection ditch to the instrument pit was blocked and so the whole pipe run was re-laid (Vinten *et al.* 1991).

4.6 MEASUREMENT OF OTHER N CYCLE PROCESSES

4.6.1 RAINFALL N INPUT

Rainfall N input was initially not measured at the experimental site, however when it became evident that there was a lack of available data on local atmospheric N inputs (Dr. D. Fowler, Institute for Terrestrial Ecology, personal communication, 1988) a rain water collector was installed.

Rain water was analysed for NO_3-N and NH_4-N (3.1.5) between 4 October, 1988 and 27 June, 1989. N inputs for the period were calculated using data from the tipping bucket rain gauges (4.5.1). No estimates of dry deposition were attempted.

Rainfall N inputs are generally highly correlated with total rainfall volume (Roberts 1987) and it seemed reasonable to assume that the rainfall N input between October 1988 and June 1989 could be extrapolated to the whole experimental period.

4.6.2 N_2 FIXATION AND LEGUME N INPUT

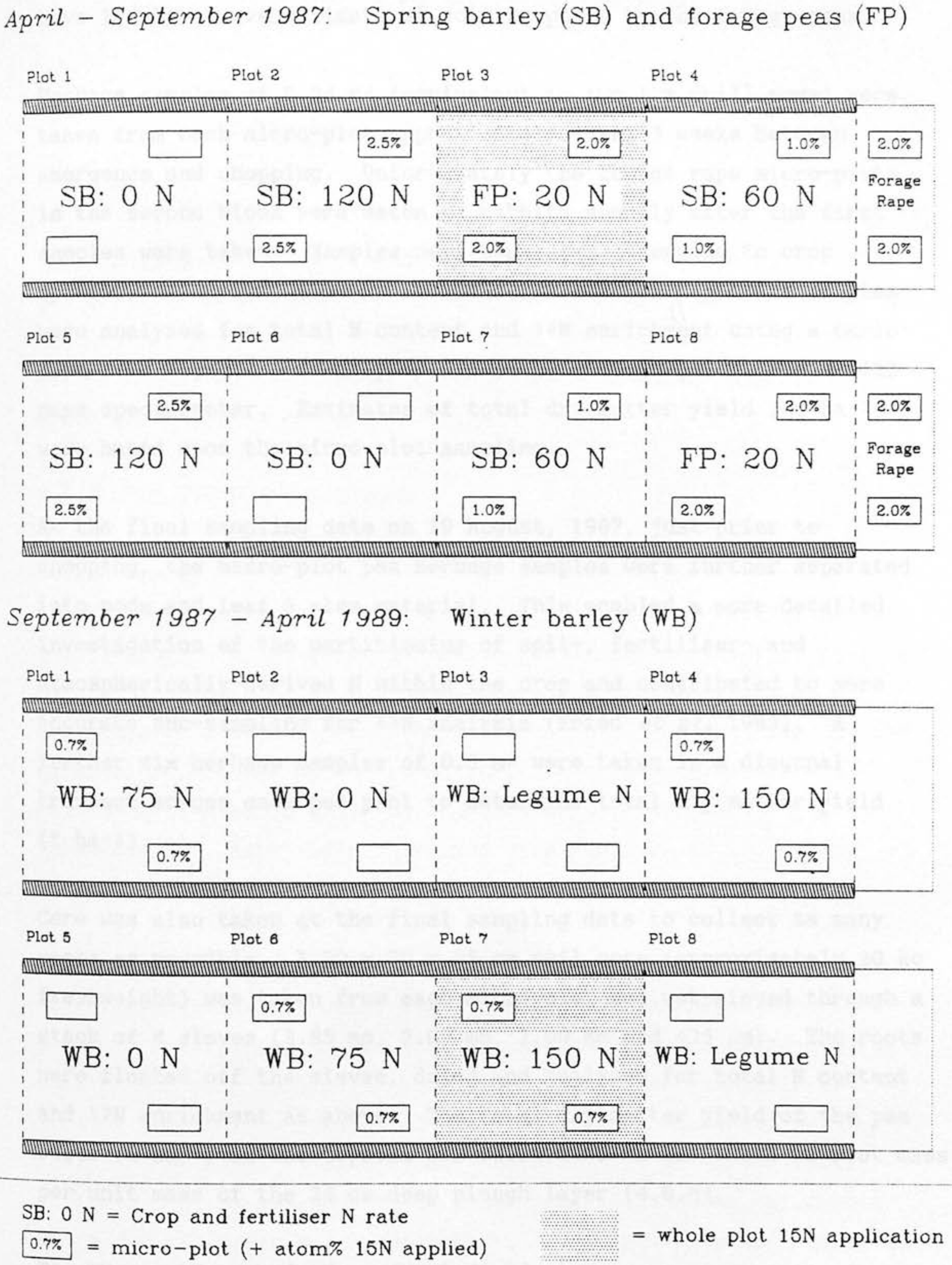
A ^{15}N-labelled fertiliser dilution technique was used to estimate symbiotic N_2 fixation in the forage peas prior to their incorporation as a green manure. In recognition of the limitations of the technique (2.2) an alternative labelling technique, such as the addition of labelled organic residues was considered. However, since fertiliser N was specifically applied to the field-grown forage peas as an agronomic measure to encourage early growth and nodulation (4.4), it was not appropriate to use an alternative labelling technique for the estimation of N_2 fixation. Instead it was decided to use two reference crops: forage rape (*Brassica napus*) and the spring barley being grown in association with the forage peas.

Forage rape is a quick growing, leafy forage crop likely to have a similar N uptake profile to forage peas. Barley is not generally considered to be a good reference crop (*e.g.* Witty 1983, Jensen 1986a), although it is still often used as such. In this work however it was felt that its disadvantages as a reference crop would be offset by the benefits of having the legume and reference crop situated as closely together as possible (Reichardt *et al.* 1987). Several authors (*e.g.* Danso *et al.* 1987, Chalk 1985) have reported that there is no evidence of N transfer between arable legumes and inter-cropped cereals, so this was not considered a potential problem when using the barley in this manner.

The forage rape, cv. Crack (Twyford Seeds Ltd., Banbury), was sown by hand on April 29, 1987 in four 2.0x3.0 m plots in the guard areas of the blocks (Figure 4.5). The seed rate was 10 g per plot, equivalent to approximately 17 kg ha^{-1}. This was twice the recommended seed rate (SAC 1987b), to allow for reduced viability due to the age of the seed. The guard areas had been drilled with spring barley prior to sowing the forage rape and on emergence the barley in the forage rape plots was thinned out to a similar plant population to that present in the forage pea plots. Thinning out was achieved by simply 'wiping' the excess barley plants with a 20% glyphosate solution.

Micro-plots (1.5x2.0 m) were marked out within each small forage rape plot, and within the pea plots (Figure 4.5). Micro-plots in the pea plots were covered during the whole plot applications of 'starter' N (4.4, 4.5.3). An aqueous solution (≈500 ml) of 2.0 atom% $^{15}NH_4{}^{15}NO_3$ was applied to all the micro-plots on May 22, 1987, at a rate equivalent to 20 kg N ha^{-1}. Fertiliser solutions were prepared as for the whole plot applications (4.5.3).

Figure 4.5: Summary of fertiliser N application and the use of micro-plots at the Glencorse experimental site during April-September 1987 and October 1988-April 1989



A higher rate of fertiliser N (*i.e.* the "A" value modification) was not used on the forage rape, since there is evidence of increased fertiliser rates enhancing soil N uptake on imperfectly drained till soils such as the Winton series (Smith *et al.* 1984). This would have led to an overestimate of soil N uptake by the forage peas.

Herbage samples of 0.24 m^2 (equivalent to two 1 m drill rows) were taken from each micro-plot approximately every 3 weeks between emergence and chopping. Unfortunately the forage rape micro-plots in the second block were eaten by rabbits shortly after the first samples were taken. Samples were separated according to crop species, dried for 24 hours at 105°C, milled and duplicate samples were analysed for total N content and ^{15}N enrichment using a Carlo-Erba 1400 Automatic N Analyser linked to a VG-Isogas Micromass 622 mass spectrometer. Estimates of total dry matter yield (kg ha^{-1}) were based upon the micro-plot sampling.

At the final sampling date on 19 August, 1987, just prior to chopping, the micro-plot pea herbage samples were further separated into pods and leaf & stem material. This enabled a more detailed investigation of the partitioning of soil-, fertiliser- and atmospherically-derived N within the crop and contributed to more accurate sub-sampling for ^{15}N analysis (Fried *et al.* 1983). A further six herbage samples of 0.5 m^2 were taken in a diagonal transect across each pea plot to determine total dry matter yield (t ha^{-1}).

Care was also taken at the final sampling date to collect as many roots as possible. A 20 x 20 x 25 cm soil core (approximately 20 kg freshweight) was taken from each micro-plot and wet-sieved through a stack of 4 sieves (3.35 mm, 2.00 mm, 1.00 mm and 425 μm). The roots were floated off the sieves, dried and analysed for total N content and ^{15}N enrichment as above. The total dry matter yield of the pea roots (t ha^{-1}) in the topsoil was calculated on the basis of root mass per unit mass of the 24 cm deep plough layer (4.6.4).

The $\%N_{dfa}$, $\%N_{dff}$ and $\%N_{dfs}$ for individual crop species, crop fractions and sampling dates were estimated according to equations

3, 5 and 6 respectively (2.2):

$$\%N_{dfa} = 1 - \left[\frac{\text{atom\% }^{15}\text{N excess (legume material)}}{\text{atom\% }^{15}\text{N excess (reference crop)}}\right] \times 100 \qquad (3)$$

$$\%N_{dff} = \frac{\text{atom\% }^{15}\text{N excess (plant)}}{\text{atom\% }^{15}\text{N excess (fertiliser)}} \times 100 \qquad (5)$$

$$\%N_{dfs} = 100 - (\%N_{dff} + \%N_{dfa}) \qquad (6)$$

N_{dfa}, N_{dff} and N_{dfs} were calculated as kg N ha^{-1} using the estimates of total legume N content derived from the %N contents measured in micro-plots and the available estimates of total dry matter yield. All calculations were made on a micro-plot basis and appropriate means (*n* = 4) were prepared.

4.6.3 CROP YIELD AND N UPTAKE

Spring barley

Micro-plots (1.5x2.0 m) were marked out within each spring barley plot (Figure 4.5) and covered during fertiliser application on 1 May, 1987. Immediately following the whole plot fertiliser application an aqueous solution (≈500 ml) of $^{15}NH_4{}^{15}NO_3$ was applied at an appropriate rate to the micro-plots in the recommended and reduced N treatments. ^{15}N enrichments were 1.0 and 2.5 atom% respectively. Fertiliser solutions were prepared as previously described (4.5.3).

Plant samples were taken from the micro-plots in all spring barley plots at anthesis (31 July, 1987) and harvest (10 September, 1987). Two 1 m rows (0.24 m^2 area) were harvested from each micro-plot by cutting the crop with scissors within a few mm of ground level. No attempt was made to sample root material. Plant material was analysed for total N content and ^{15}N enrichment as for the peas (4.6.2). At final harvest the plant material was separated into grain and straw for yield determination and analysis. An additional twenty-one 1.0 m rows (2.52 m^2 area) were taken from a diagonal transect across the plots for a more accurate estimate of total dry matter yield and N uptake. All grain yields were expressed at 15% moisture content.

$\%N_{dff}$ was estimated according to equation (5) and $\%N_{dfs}$ as follows:

$$\%N_{dfs} = 100 - \%N_{dff} \tag{12}$$

All yield and N uptake data were prepared as treatment means (n = 4) for each harvest date.

Winter barley
New micro-plots were marked out in the winter barley shortly after establishment (Figure 4.5) and crop N uptake was regularly measured from all plots by harvesting and analysis as in the spring barley. At the time of fertiliser application the micro-plots were covered and then sprayed with an aqueous solution of 0.7 atom% $^{15}NH_4{}^{15}NO_3$ at an appropriate rate. Final harvest (26 July, 1988) was conducted in a similar manner to that of the spring barley.

During July 1988 the plots suffered some damage by rooks and one micro-plot from the recommended N treatment in plot 4 was lost.

Winter rye
Plant samples were taken from the winter rye plots on 15 November, 1988, and 13 March, 1989, just prior to ploughing of the plots. On each occasion four 0.25 m^2 areas were cut in a diagonal transect across the plots. Total N uptake was determined as previously described.

4.6.4 SOIL MINERAL N

Following incorporation of the forage peas in August 1987 soil mineral N levels were measured on a regular basis from the micro-plot areas marked out in 4.6.3. The first samples were taken from the zero and legume N plots on 4 September at the time of green manure incorporation and from all other plots on 28 September. Measurements continued until April 1989 and were taken most frequently (≈once a month) from the zero and legume N treatments, plus Plots 1 and 6 following establishment of the winter rye (4.4).

At sampling four 20 cm deep cores were taken from each micro-plot. The cores for each micro-plot were bulked, sieved and duplicate samples were extracted by shaking for 1 hour with 1M KCl. Extracts were analysed for NO_3-N and NH_4-N by the same procedure as for the water samples (3.1.5) and, following correction for soil moisture content, were expressed as mg N kg^{-1} soil.

Further correction enabled the estimation of kg mineral N ha^{-1}, although the absolute figures should be treated cautiously. Table 4.1 suggests that the typical depth of the plough layer at Glencorse was 24 cm with a bulk density of 1.19 g cm^{-3}. The mass of topsoil per hectare was therefore 2 856 t. Assuming that sampling to 20 cm provided an unbiased estimate of mineral N levels in the whole of the topsoil, a correction factor of 2.856 was used to convert mg N kg^{-1} to kg N ha^{-1}.

On 25 May, 1988, four soil cores were taken to a depth of 40-50 cm from Plot 7 using a 75 mm diameter lined corer. The cores were divided into 10 cm segments and a sub-sample of each of segment was extracted with 1M KCl as above. After analysis for NO_3-N and NH_4-N content (3.1.5), the extracts were steam distilled (4.5.3) and the distillate for each core depth bulked before ^{15}N analysis. $^{15}NO_3$-N and $^{15}NH_4$-N enrichments were determined separately for the bulked 0-10 cm segments, but not for the other depths. $\%N_{dff}$ remaining in the soil was calculated as follows:

$$\%N_{dff} = \frac{\text{atom\% } ^{15}\text{N excess (soil extract)}}{\text{atom\% } ^{15}\text{N excess (fertiliser)}} \times 100 \qquad (13)$$

4.6.5 DENITRIFICATION LOSSES

Denitrification losses from the plots were monitored at weekly intervals from the end of September 1987 until November 1988 using an adaptation of the "acetylene inhibition technique" described by Ryden *et al.* (1979).

Plate 4.5: Use of sealed chamber in an adapted "acetylene inhibition technique" for measuring denitrification losses in soil

One 0.5 m^2 sealed chamber was installed (Plate 4.5) in each plot on 23 September, 1987 after all the plots had been ploughed and was subsequently re-installed after each field operation. The chambers were similar in design to those used by other workers (*e.g.* Matthias *et al.* 1980) and included a vent to allow the transmission of atmospheric pressure fluctuations to within the chamber (Hutchinson and Mosier 1981). Once a week the chambers were flooded with acetylene, which was left to diffuse into the soil for 24 hours before duplicate gas samples were taken. N_2O concentrations were measured by gas chromatography using an electron capture detector (Pye Unicam Series 104 chromatograph).

Relatively low concentrations of acetylene (0.01 m^3 m^{-3}) in the soil atmosphere inhibit the microbial reduction of N_2O to molecular N_2 and Ryden *et al.* (1979) assumed that the N_2O flux from a soil in the presence of acetylene is representative of the total denitrification flux. It is unlikely however that acetylene inhibition was particularly successful on the Winton soil series because of the difficulties of establishing adequate acetylene concentrations throughout such a heavily aggregated soil (Arah *et al.* 1991). Nonetheless, the measurement of N_2O emissions by the adapted technique used was very straightforward and provided an indication of the occurrence and extent of total denitrification activity in the different treatments.

In most soils N_2O flux displays a log-normal spatial distribution (Folorunso and Rolston 1984). Weekly treatment means were therefore calculated using a geometric rather than arithmetic mean.

4.6.6 N RELEASE FROM THE INCORPORATED LEGUMINOUS GREEN MANURE

A number of techniques have been used by researchers to measure, or obtain an index, of N mineralisation rates. However, there is still no reference method which is known to accurately measure N mineralisation rates under field conditions (Raison *et al.* 1987).

Following the initial shallow incorporation of the green manure on 4 September, 1987 (4.4), two approaches were adopted to assess the pattern of N mineralisation in the zero and legume N treatments:

Method 1 - 'N recovery'

Meisinger (1984) suggested that the underlying principle for evaluating available N in plant-soil systems should be the N balance (discussed further in 6.3). The potential application of N balance principles varies greatly depending upon the resources available.

On the hydrologically isolated plots at Glencorse it was proposed to estimate N mineralisation (kg N ha^{-1}) from the cumulative 'recovery' of N in leaching losses, crop N content and soil mineral N pools at regular intervals after incorporation. The estimates were corrected to account for the rainfall N input, but no correction was made for N supply from seed or for N recovery in roots.

N uptake by roots is rarely estimated in N studies because of the inherent difficulties in accurately sampling and extracting root biomass from soil cores, particularly in crops with relatively fine, fibrous root systems such as cereals. For the purposes of estimating N mineralisation here, it was simply assumed that root N uptake was equivalent to seed N input throughout the crop's growth period.

This was probably a major underestimate of root N content, but root growth was likely to be severely restricted in this soil type (Holmes 1976) and there was no information available on how the root biomass of the crop would alter with time (Hansson and Steen 1984) or interact with available N (Welbank and Williams 1968) under these conditions.

Total N mineralisation, TN_{min} (kg N ha^{-1}), since September 4, 1987 was calculated as follows:

$$TN_{min(t)} = TN_{l(t)} + TN_{c(t)} + \left[N_{(t)} - N_{(0)}\right] - TN_{r(t)} \qquad (14)$$

where:

$TN_{l(t)}$ = cumulative N leaching losses (kg N ha^{-1}) at time t;

$TN_{c(t)}$ = crop N content (kg N ha^{-1}) at time t;

$N_{(t)}$ = soil mineral N content (kg N ha^{-1}) at time t;

$N_{(0)}$ = soil mineral N content (kg N ha^{-1}) at time 0;

$TN_{r(t)}$ = cumulative rainfall N input (kg N ha^{-1}) at time t.

Net mineralisation, N_{min} (kg N ha^{-1}), for a given time period (t-1 to t) was estimated by:

$$N_{min} = TN_{min(t)} - TN_{min(t-1)} \qquad (15)$$

Method 2 - Soil incubation

Field incubated soil cores have been used in a number of N mineralisation studies (*e.g.* Smith *et al.* 1977, Westermann and Crothers 1980, Raison *et al.* 1987), including under local conditions (Rees 1989). In this work the use of a sequential soil core incubation technique was used to give an independent estimate of N mineralisation for comparison with Method 1.

Starting on 20 October, 1987 and then on subsequent soil sampling dates (4.6.4), four 20 cm soil cores were taken from each microplot, placed as intact as possible into polythene bags and returned to their respective holes in the micro-plot (Rees 1989). Simple household bags were used with a small perforation placed in each bag to limit the development of anaerobic conditions. A plastic cover was placed over each core to prevent water entering. Following field incubation (≈one month) the cores from each micro-plot were removed, bulked and extracted for NH_4-N and NO_3-N (4.6.4). After correction for moisture content and bulk density the results were expressed as kg N ha^{-1} (4.6.4) and prepared as a mean treatment value (n = 4).

Field incubations began 46 days after incorporation of the green manure and continued in parallel to the measurements being made for Method 1 above. Net mineralisation, N^*_{min} (kg N ha^{-1}), for each treatment and incubation period was estimated using a similar rationale to that proposed by Raison *et al.* (1987) as follows:

$$N^*_{min} = N_{b(t)} - N_{s(t-1)} \tag{16}$$

where:

$N_{b(t)}$ = mean mineral N content (kg N ha^{-1}) of the soil cores in the plastic bags at the end of incubation, time t;

$N_{s(t-1)}$ = soil mineral N content (kg N ha^{-1}) measured at the start of incubation, time t-1.

Total N mineralisation, TN^*_{min} (kg N ha^{-1}), from the start of incubations to time t, was calculated as the sum of N^*_{min} values for all preceding periods.

Due to experimental problems, no zero N incubation was made between March and April 1988. It was thus decided to estimate N accumulation in the zero N treatment during this period using a simple N mineralisation model. Addiscott (1983) suggested that net N mineralisation (N_t) occurring during time, t could be estimated using the following zero-order equation:

$$N_t = k_t \tag{17}$$

where the temperature dependence of k was best expressed by an Arrhenius-type relationship with absolute temperature (T):

$$k = Ae^{-B/T} \tag{18}$$

The constant values of A and B for the zero N estimate were assumed to be the same as those calculated by Addiscott (1983) for a general arable soil. Daily soil temperature for the period March to April 1988 was available from the nearby Bush House weather station (5.1). The soil temperature at 20 cm depth was used in order to reduce the error due to diurnal variation (Russell 1973), whilst still maintaining some relevance to the 0-20 cm sampling depth.

Net 'apparent' mineralisation

The net 'apparent' mineralisation of legume material for a given time period was derived from equations 15 (Method 1) and 16 (Method 2), as follows:

$$L_{min(t)} = N_{lmin(t)} - N_{zmin(t)} \tag{19}$$

$$L^*_{min(t)} = N^*_{lmin(t)} - N^*_{zmin(t)} \tag{20}$$

where:

l and z refer to the legume and zero N treatments respectively.

4.7 STATISTICS

Extensive statistical analysis was not justified on this work because of the predominantly small sample sizes (maximum $n = 4$). Snedecor and Cochran (1967) stressed that significance testing on small samples can be particularly weak and suggested that greater emphasis be placed upon the simple interpretation of trends etc. Other authors (*e.g.* Downie and Heath 1974) have also suggested that in many situations an indication of the possible magnitude of an effect may be a more helpful way of reporting results than significance testing. Nonetheless, it is still useful to consider the differences between means and be able to make some judgement as to whether these differences are real or not.

The majority of data presented for the Glencorse experiment has been prepared as treatment means and is reported with standard errors of these means. Where appropriate the Studentised Range test has been used to test the significance ($p = 0.05$) of differences between means (Snedecor and Cochran 1967).

The Studentised Range test involves the calculation of a Least Significant Range (LSR). This is a modification of the commonly used Least Significant Difference (LSD) which is prone to Type I statistical errors *i.e.* judging too many differences to be significant (Downie and Heath 1974). LSRs involve the use of tabulated Q-values rather than t-values and provide a more rigorous test of significance by effectively imposing wider confidence intervals upon the treatment means.

5.1 METEOROLOGICAL DATA

Meteorological data for April 1987 to March/April 1989 was available from a weather station at Edinburgh University's Bush House, about 1 km north-east of the experimental site, and from the Meteorological Office (Bracknell, Berks.). Quarterly rainfall and potential evaporation measurements (Table 5.1), along with soil temperature data (Figure 5.1), are included here to provide an overview of the environmental conditions prevailing during the experimental work.

Rainfall was measured at the experimental site between September 1987 and April 1989 and correlated well with Bush House data over the same period. The on-site measurements were specifically used for investigation of the drainflow recovery from the experimental plots (5.3.1) and the estimation of rainfall N inputs (5.3.3).

5.2 PRELIMINARY EXPERIMENTAL PERIOD: APRIL - SEPTEMBER 1987

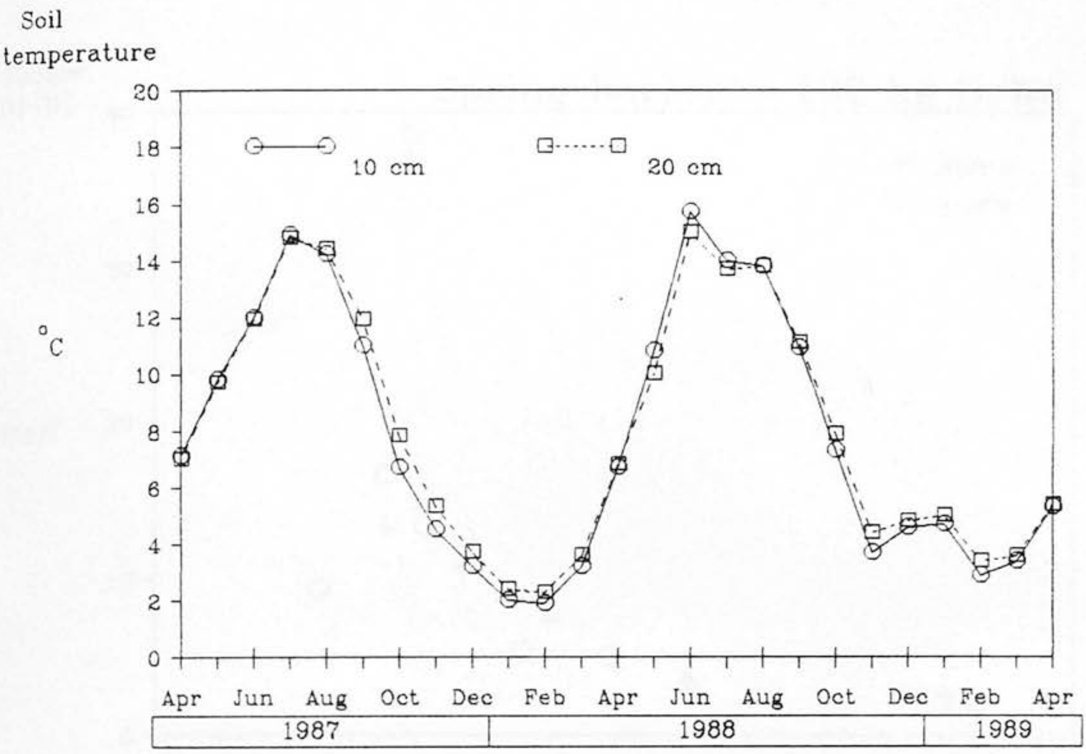
5.2.1 NITRATE LEACHING

No detailed measurements of NO_3-N leaching were made during the preliminary experimental period, but the NO_3-N concentrations of drainflow spot samples are included in Figures 5.2a and 5.2b.

Table 5.1: Quarterly rainfall and potential evaporation (mm) from April 1987 - March 1989 (including long term averages)

Period:	Rainfall (mm)	Potential evap., ET_p (mm)	Long term averages: Rainfall	ET_p
Apr-Jun 1987	221.8	170.2	178	194
Jul-Sep	252.0	174.9	269	179
Oct-Dec	239.9	27.0	238	32
Jan-Mar 1988	238.2	55.5	181	47
Apr-Jun	138.3	183.5	178	194
Jul-Sep	340.4	175.2	269	179
Oct-Dec	178.0	28.2	238	32
Jan-Mar 1989	268.5	62.8	181	47
Total:	1877.1	877.3	1732	904

Figure 5.1: Mean monthly soil temperatures at 10 and 20 cm depths during the experimental period, Bush House, Midlothian.



The variability in drainflow from the individual plots (4.5.4) meant that spot samples were not consistently taken from all the plots (*e.g.* only one spot sample was taken from plot 3 all summer). However, some basic trends could be identified.

The temporal distribution of NO_3-N concentrations was similar in all treatments. Maximum concentrations were in June, with a decline towards September. The highest NO3-N concentrations (Figure 5.2a) were measured from the recommended N (37.9 mg l^{-1}) and reduced N (21.4 mg l^{-1}) treatments on 8 June following 28.5 mm rainfall in the preceding 48 hours. This was not the first major rainfall event following fertiliser N application (*e.g.* 13.1 mm was recorded on 29 May), but it was the first to coincide with a spot sample. The peaks in NO_3-N from the zero N treatment and from the forage peas were not so clearly defined, but were smaller (13.0 - 14.0 mg NO_3-N l^{-1}) (Figure 5.2b).

Figure 5.2a: NO_3-N concentrations of spot samples taken from the drainflow of plots 2, 4, 5 and 7 during the preliminary experimental period (May - September 1987).

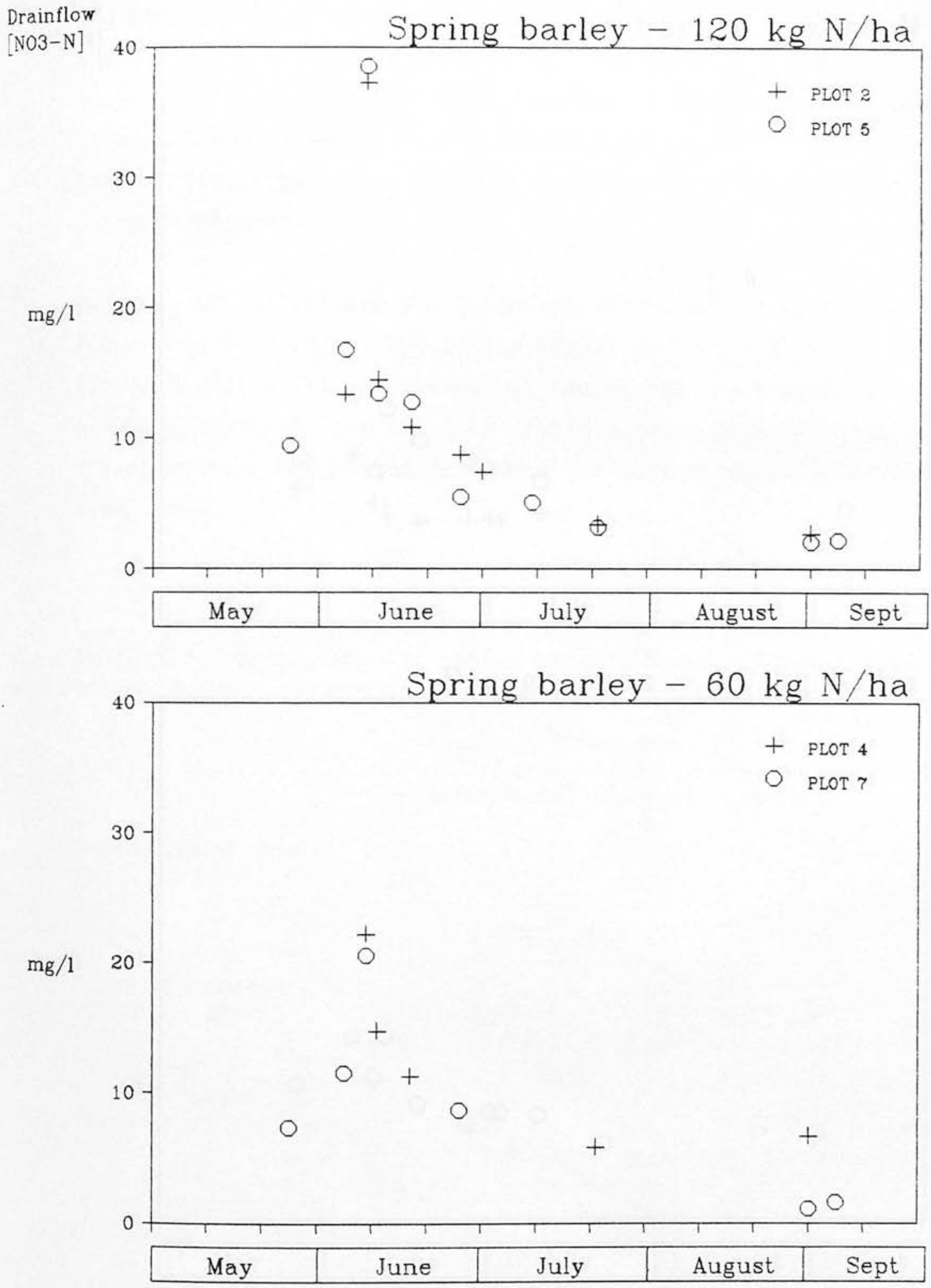
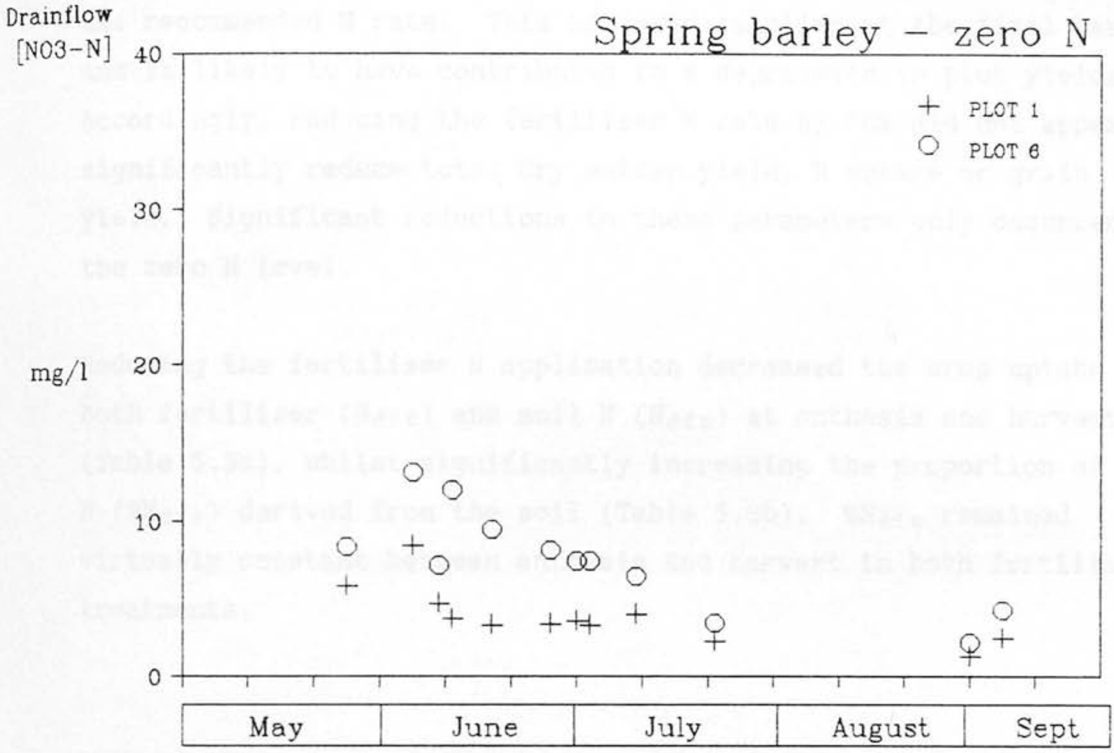
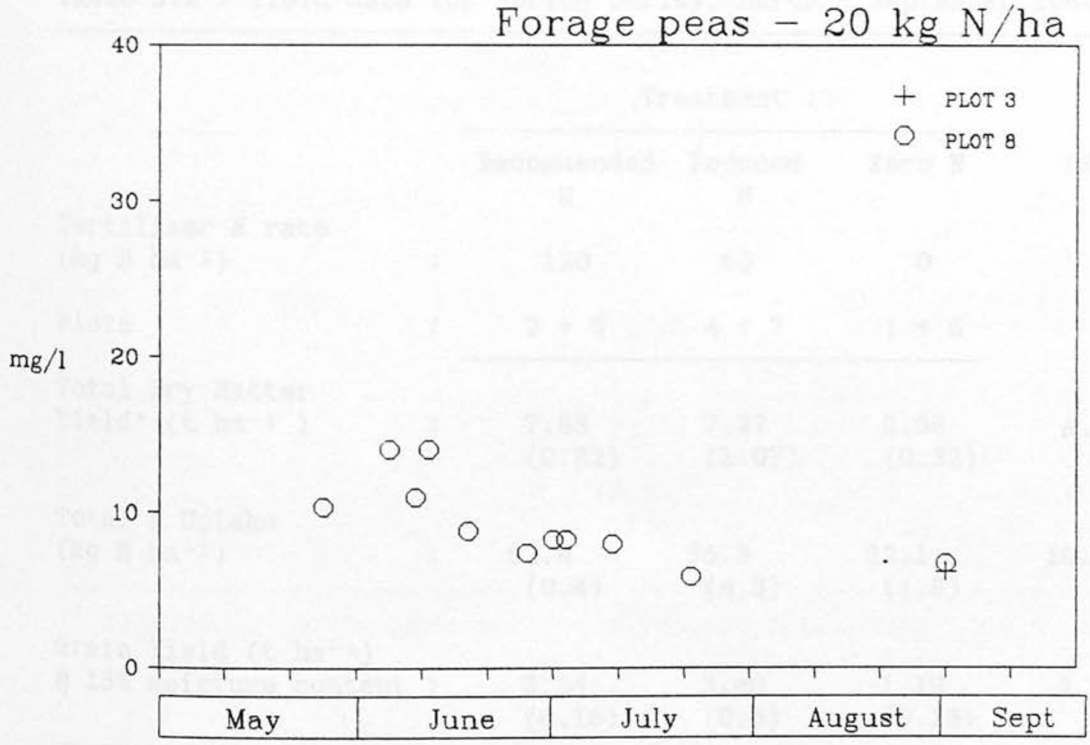


Figure 5.2b: NO_3-N concentrations of spot samples taken from the drainflow of plots 1, 3, 6 and 8 during the preliminary experimental period (May - September 1987).





5.2.2 SPRING BARLEY CROP

Mean yield data for the spring barley crop is summarised in Table 5.2. Dry matter yield, crop N uptake and grain yield were highest in the recommended N treatment and declined with decreasing fertiliser N application. There was some lodging of the crop at the recommended N rate. This hindered sampling at the final harvest and is likely to have contributed to a depression in plot yields. Accordingly, reducing the fertiliser N rate by 50% did not appear to significantly reduce total dry matter yield, N uptake or grain yield. Significant reductions in these parameters only occurred at the zero N level.

Reducing the fertiliser N application decreased the crop uptake of both fertiliser (N_{dff}) and soil N (N_{dfs}) at anthesis and harvest (Table 5.3a), whilst significantly increasing the proportion of crop N (%N_{dfs}) derived from the soil (Table 5.3b). %N_{dfs} remained virtually constant between anthesis and harvest in both fertiliser N treatments.

Table 5.2 : Yield data for spring barley, March - September 1987

		Treatment :			
		Recommended N	Reduced N	Zero N	LSR
Fertiliser N rate (kg N ha^{-1})	:	120	60	0	
Plots	:	2 + 5	4 + 7	1 + 6	
Total Dry Matter Yield* (t ha^{-1})	:	7.83 (0.82)	7.27 (1.07)	2.58 (0.32)	**4.73**
Total N Uptake (kg N ha^{-1})	:	62.4 (0.4)	56.3 (4.3)	22.1 (1.8)	**10.7**
Grain Yield (t ha^{-1}) @ 15% moisture content	:	3.54 (0.16)	3.40 (0.5)	1.19 (0.13)	**1.23**

() Figures in parentheses are standard errors of means (n = 4, except * where n = 2)

Table 5.3a: Uptake of soil and fertiliser N (kg ha^{-1}) in above-ground plant material of spring barley crop at anthesis (31/7/87) and final harvest (10/9/87)

	ANTHESIS				HARVEST			
Fertiliser N (kg ha^{-1}):	120	60	0	LSR	120	60	0	LSR
N_{dfs}	38.1 (2.2)	34.7 (2.1)	19.2 (0.1)	6.9	32.4 (0.7)	37.3 (2.4)	22.1 (1.8)	7.0
N_{dff}	35.6 (3.3)	18.1 (2.3)	-	9.8	30.1 (1.0)	19.0 (2.4)	-	6.4
% Rec. of Fert. N	29.7 (2.8)	30.2 (3.9)	-	11.8	25.1 (0.8)	31.7 (4.0)	-	10.0

() Figures in parentheses are standard errors of means (n = 4)

Table 5.3b: Percentage of N in above-ground plant material of spring barley crop derived from soil (%N_{dfs}) at anthesis (31/7/91) and final harvest (10/9/87)

Fertiliser N (kg ha^{-1}) :	120	60	LSR
Shoot at anthesis :	51.7 (1.5)	65.8 (2.6)	7.3
Grain :	51.6 (1.4)	66.0 (2.5)	7.0
Straw :	52.3 (1.4)	66.8 (2.0)	6.0

() Figures in parentheses are standard errors of means (n = 4)

At harvest and anthesis the zero N treatment showed a significantly lower absolute uptake of soil N than the two fertiliser treatments (Table 5.3a).

The % recovery of applied ^{15}N-labelled fertiliser in the harvested crop remained within the range of 25-32% for both fertiliser treatments and sample dates (Table 5.3a). It is likely that the recovery figure for the recommended N treatment at harvest is an underestimate due to the sampling problems at this time.

5.2.3 GREEN MANURE CROP

Yield and N content at the time of chopping

The total N content of the green manure crop (Plate 5.1) at the time of chopping was 358.3 kg N ha^{-1} (310.2 kg ha^{-1} above-ground and 48.1 kg ha^{-1} below-ground). This was virtually all (≈99%) derived from the forage pea material (Table 5.4) which, on a dry matter basis, consisted of 3.34 t ha^{-1} of flat pods at 3.5% N content, 6.29 t ha^{-1} of leaf and stem material at 3.0% N, and 3.27 t ha^{-1} of root material at 1.5% N. The seed rate of the intercropped barley had been very low (4.4) and at chopping the dry matter yield of the aboveground portion was only 0.24 t ha^{-1} at 1.7% N. The C:N ratio of the incorporated material was not analysed, but it was presumably of predominantly low C:N ratio.

Almost 89% of the total N content of the forage peas was estimated to be derived from symbiotic N_2 fixation *i.e.* 314.2 kg N ha^{-1} compared to 5.2 and 34.8 kg N ha^{-1} derived from fertiliser and soil respectively (Table 5.5 and 5.6). The intercropped barley contained 0.5 and 3.5 kg N ha^{-1} from fertiliser and soil N respectively (Table 5.5) and its $\%N_{dfs}$ (Table 5.6) was considerably higher than in the fertilised spring barley plots (Table 5.3b).

Total fertiliser uptake by the green manure (peas and barley) was 5.7 kg N ha^{-1} (Table 5.5) *i.e.* a fertiliser recovery of 28.5% which was similar to the recoveries found in the spring barley crops (Table 5.3a).

Plate 5.1: Green manure crop prior to chopping and incorporation on the experimental plots at East Flotterstone Field

Table 5.4: Total dry matter yield (t ha^{-1}) and N content (kg ha^{-1}) of green manure crop at the time of chopping (19/8/87)

	DM yield (t ha^{-1})	%N	N content (kg ha^{-1})
Forage peas - pods :	3.34 (0.25)	3.5	115.4 (9.5)
- leaf & stems :	6.29 (0.25)	3.0	190.7 (9.0)
- roots :	3.27 (0.38)	1.5	48.1 (4.2)
TOTAL - peas :	**12.90 (0.39)**		**354.3 (5.8)**
Spring barley - shoot :	0.24 (0.08)	1.7	4.0 (1.1)
TOTAL - green manure :	**13.14 (0.40)**		**358.3 (6.8)**

() Figures in parentheses are standard errors of means (n = 4)

Table 5.5: Origin of N (kg ha^{-1}) in the green manure crop at the time of chopping (19/8/87)

	N_{dfa}	N_{dff}	N_{dfs}
Forage peas - pods :	106.5 (10.7)	1.2 (0.2)	7.7 (1.3)
- leaf & stems :	165.6 (3.7)	3.3 (0.7)	21.8 (5.4)
- roots :	42.1 (4.4)	0.7 (0.2)	5.3 (2.6)
TOTAL - peas :	**314.2 (11.6)**	**5.2 (0.7)**	**34.8 (7.8)**
Spring barley - aboveground :	-	0.5 (0.1)	3.5 (1.0)
TOTAL - green manure :	**314.2 (11.6)**	**5.7 (0.7)**	**38.3 (8.0)**

() Figures in parentheses are standard errors of means (*n* = 4)

Table 5.6: Percentage of N in the green manure crop at the time of chopping (19/8/87) that was derived from the atmosphere, fertiliser and soil

	%N_{dfa}	%N_{dff}	%N_{dfs}
Forage peas - pods :	91.8 (2.1)	1.1 (0.2)	7.1 (1.9)
- leaf & stems :	87.2 (2.5)	1.7 (0.3)	11.1 (2.3)
- roots :	87.6 (5.2)	1.5 (0.4)	10.9 (4.8)
TOTAL - peas :	**88.7 (2.4)**	**1.5 (0.2)**	**9.8 (2.3)**
Spring barley - aboveground :	-	13.8 (1.9)	86.2 (2.7)
TOTAL - green manure :	**87.7**	**1.6**	**10.7**

() Figures in parentheses are standard errors of means (*n* = 4)

The partitioning of atmospheric-, soil- and fertiliser-derived N was similar within all the different plant parts of the forage peas (Table 5.6), with a slightly higher proportion of atmospherically-derived N in the pods at the expense of soil-derived N. The absolute quantities of N from different sources were therefore related to the total N content of the plant parts, with the greatest quantities of atmospheric- (N_{dfa}), soil- (N_{dfs}) and fertiliser-derived N (N_{dff}) found in the leaf and stems, and least in the roots (Table 5.5).

Growth and N_2 fixation pattern

Dry matter yield and N content of the green manure crop was apparently still increasing at the time of chopping in mid-August, indeed the rate of dry matter and N accumulation in the above-ground plant material during the preceding 14 days was the highest observed during the crop's entire growth period (Figure 5.3).

Figure 5.3: Accumulation of dry matter (t ha^{-1}) and crop N (kg ha^{-1}) in above-ground plant material of the green manure crop

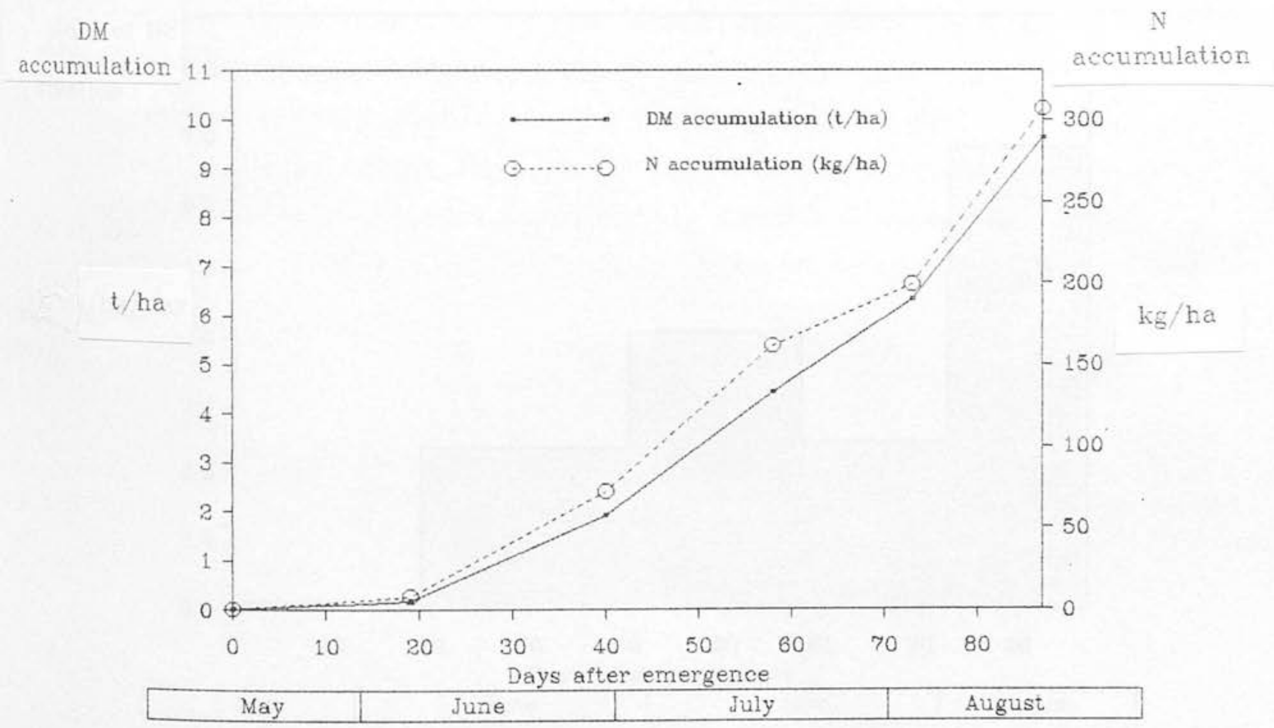


Figure 5.4: Origin of crop N in the above-ground plant material of forage peas

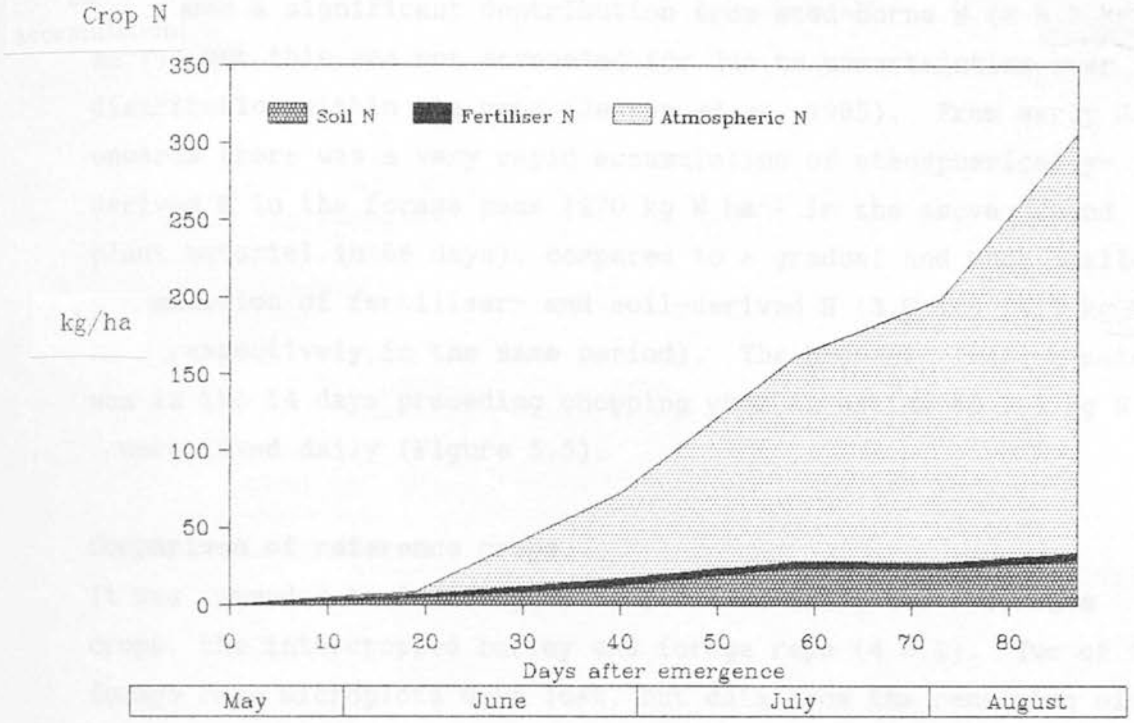
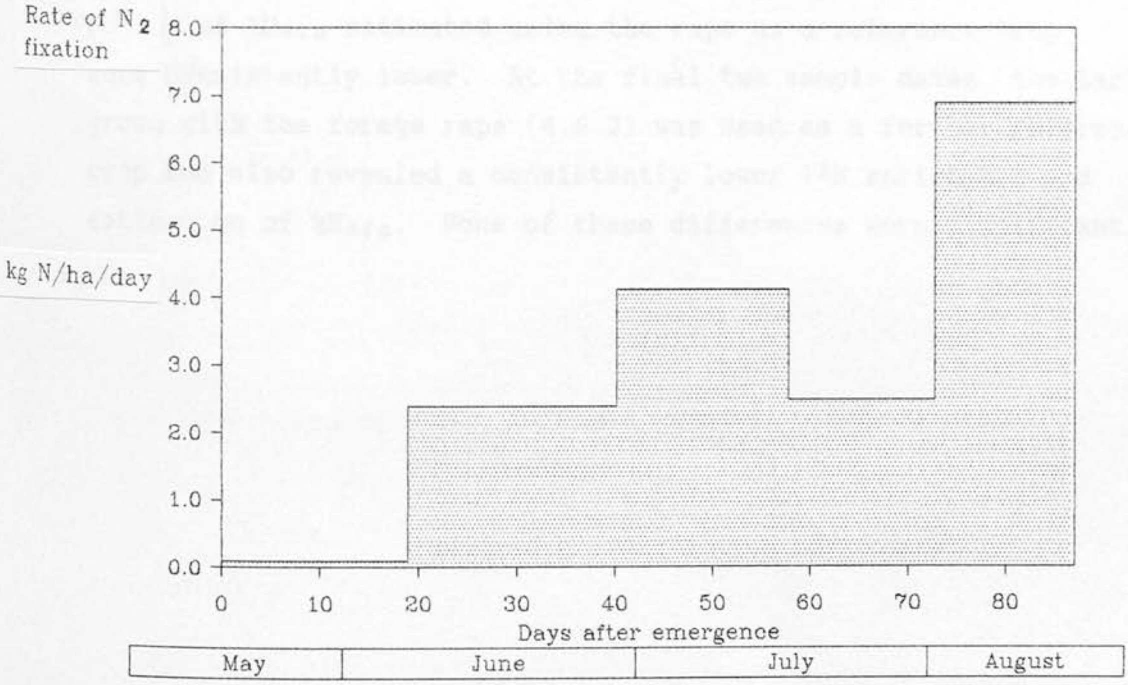


Figure 5.5: Estimated seasonal N_2 fixation rates (kg N ha^{-1} day^{-1}) in the forage peas



The accumulation of crop N from different sources in the above-ground plant material of the forage peas is shown in Figure 5.4. Up until early June (19 days after emergence) the most important sources of N were fertiliser- and soil-derived. There would also have been a significant contribution from seed-borne N ($\approx$ 8.2 kg N ha^{-1}), but this was not accounted for due to uncertainties over its distribution within the crop (Jensen *et al.* 1985). From early June onwards there was a very rapid accumulation of atmospherically-derived N in the forage peas (270 kg N ha^{-1} in the above-ground plant material in 68 days), compared to a gradual and much smaller accumulation of fertiliser- and soil-derived N (3.0 and 24.9 kg N ha^{-1} respectively in the same period). The highest fixation rate was in the 14 days preceding chopping when an estimated 7.2 kg N ha^{-1} were fixed daily (Figure 5.5).

Comparison of reference crops

It was intended to investigate N_2 fixation using two reference crops, the intercropped barley and forage rape (4.6.2). Two of the forage rape microplots were lost, but data from the remaining micro-plots are summarised in Table 5.7.

Excepting the first sample date, the ^{15}N enrichment of the forage rape was lower than that of the intercropped barley and accordingly values of $\%N_{dfa}$ estimated using the rape as a reference crop were consistently lower. At the final two sample dates, the barley grown with the forage rape (4.6.2) was used as a further reference crop and also revealed a consistently lower ^{15}N enrichment and estimation of $\%N_{dfa}$. None of these differences were significant.

Table 5.7: Estimation of $\%N_{dfa}$ in forage pea crop using three different reference crops

Sample date:	Barley* (with peas) ^{15}N atm% xs	Barley* (with peas) $\%N_{dfa}$	Forage rape ^{15}N atm% xs	Forage rape $\%N_{dfa}$	Barley (with rape) ^{15}N atm% xs	Barley (with rape) $\%N_{dfa}$	LSR
10/6/87	*0.4086*	26.3	*0.5187*	35.9	–	–	36.5
2/7/87	*0.3928*	76.4	*0.3338*	66.1	–	–	24.0
20/7/87	*0.2573*	82.6	*0.2119*	79.41	–	–	16.2
4/8/87	*0.2471*	86.0	*0.1690*	80.0	*0.1534*	78.0	12.8
18/8/87 PODS		91.8		85.7		86.2	11.7
LEAF & STEM	*0.2304*	87.2	*0.1490*	77.3	*0.1535*	77.8	11.4
ROOTS		87.7		86.8		87.1	23.8

Means calculated with $n = 2$, except * where $n = 4$

5.3 MAIN EXPERIMENTAL PERIOD: SEPTEMBER 1987 - APRIL 1989

5.3.1 PLOT HYDROLOGY

Drainflow recovery

Total rainfall during the experimental period was 1609.2 mm (total potential evaporation was 593.5 mm) and was quite evenly distributed throughout the period. Drainflow from the 8 plots was however highly variable (Figures 5.6a - 5.6d), ranging in total from 65.1 mm (Plot 3) to 1576.1 mm (Plot 5).

This variability is likely to have arisen from a number of sources including variable subsoil hydraulic conductivity and deep percolation rates (Snaebjornsson 1977); differences in actual evapotranspiration, particularly between the different N treatments in the summer (*e.g.* Dowdell *et al.* 1984); and variable precipitation across the plots. There are no data available to evaluate the significance of the first two factors, but the two tipping bucket raingauges gave similar results suggesting little variation in rainfall. There was, however, a clearly non-uniform distribution of snowfall across the plots particularly during November 1988 and February 1989.

Most of the variability arose from practical difficulties with the hydrological isolation of the plots. These problems are broadly characterised in 4.5.4.

There were also some initial problems with instrumentation at the site, notably during October 1987 when heavy rain led to flooding of the instrument pits and disabling of the tipping bucket flowmeters. This happened on several occasions before drainage from the instrument pits was improved. The problem remains evident in the data as a very marked drop in apparent % recovery of incident rainfall of some plots shortly after the start of the experiment (*e.g.* Figure 5.6a)

The only plot that consistently lost water during the entire experimental period was Plot 3 (Figure 5.6b). All attempts to

Figure 5.6a: Mean daily drainflow (mm) and cumulative % apparent recovery of incident rainfall from Plots 1 and 2 during the period 4 September, 1987 to 19, April 1989

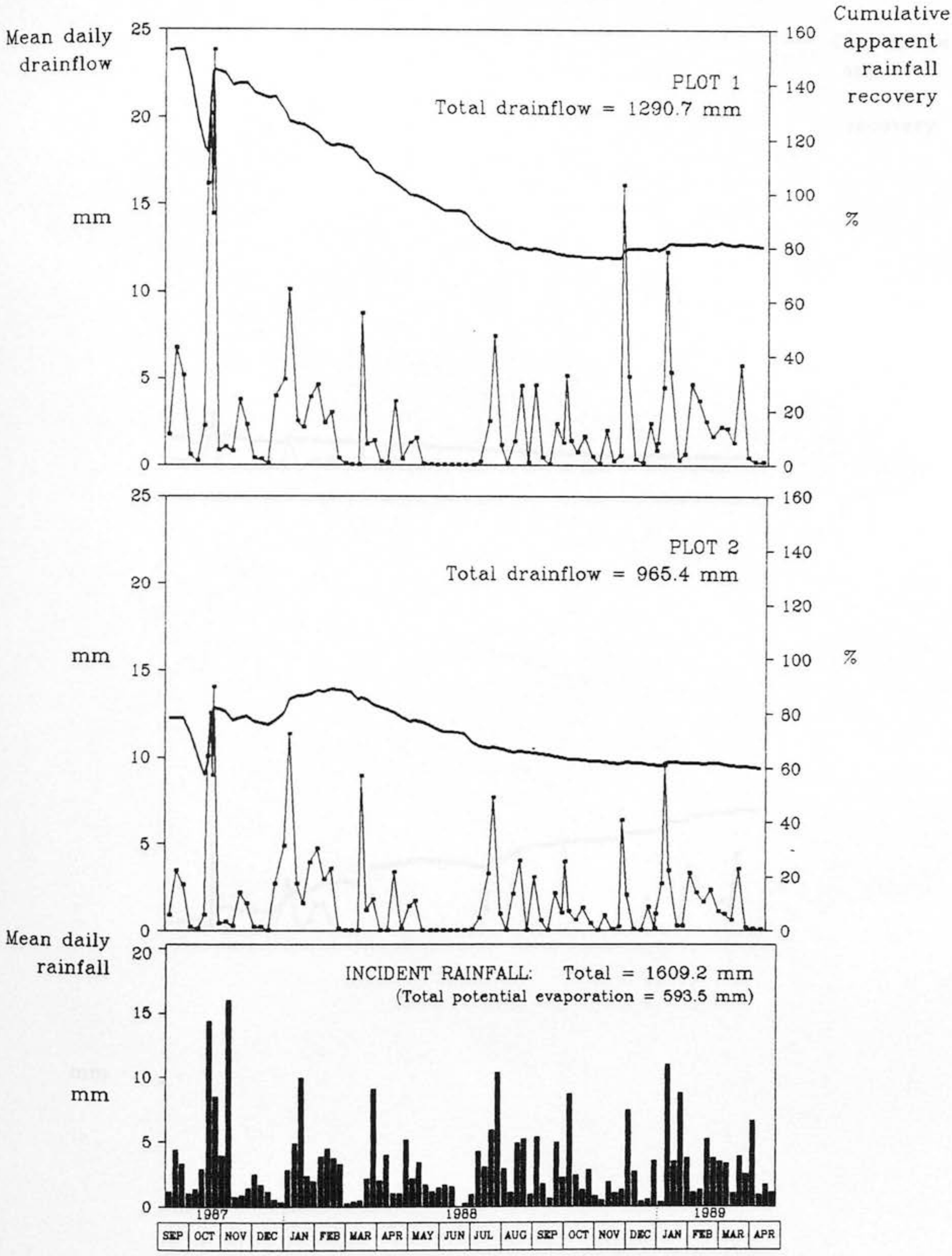


Figure 5.6b: Mean daily drainflow (mm) and cumulative % apparent recovery of incident rainfall from Plots 3 and 4 during the period 4 September, 1987 to 19, April 1989

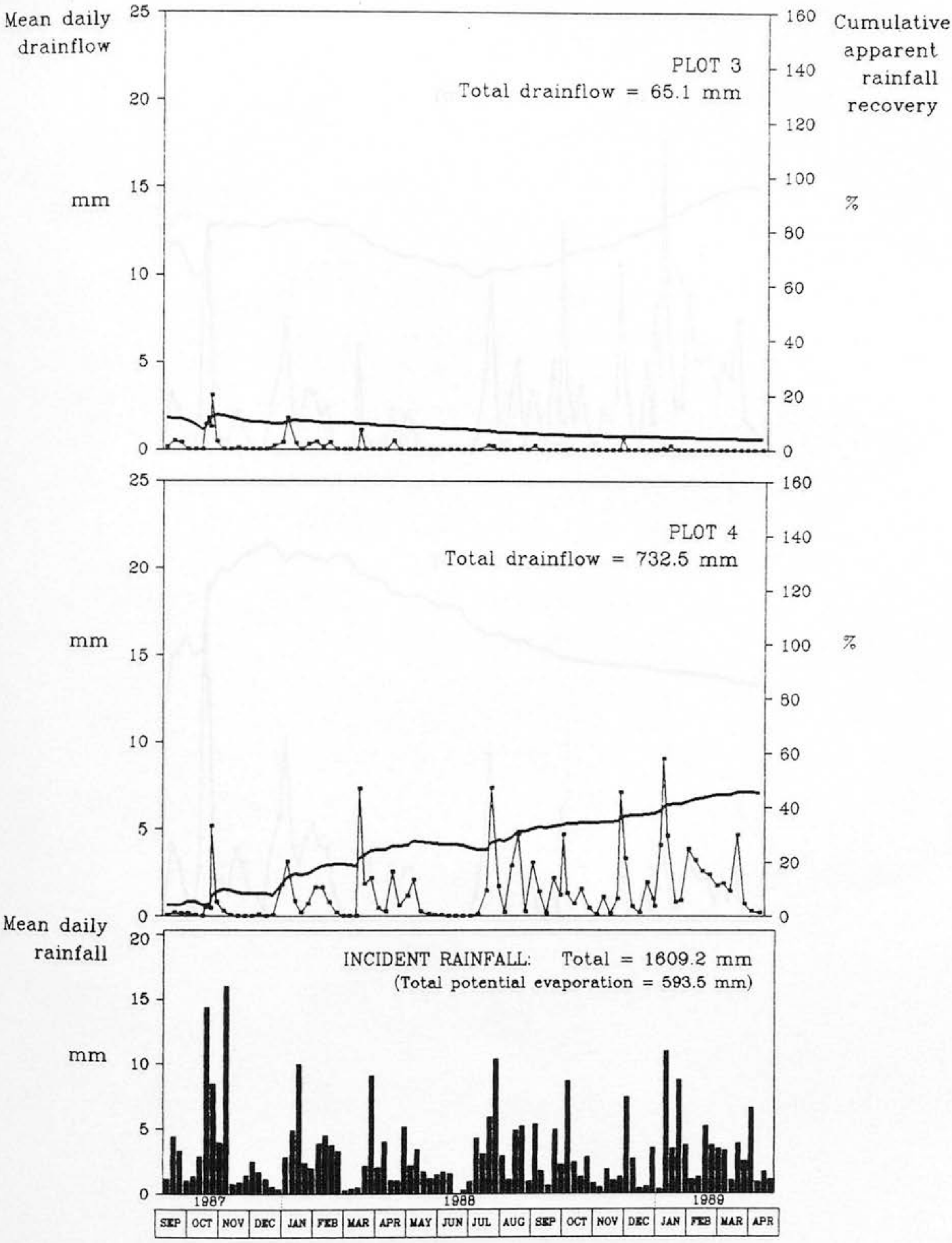


Figure 5.6c: Mean daily drainflow (mm) and cumulative % apparent recovery of incident rainfall from Plots 5 and 6 during the period 4 September, 1987 to 19, April 1989

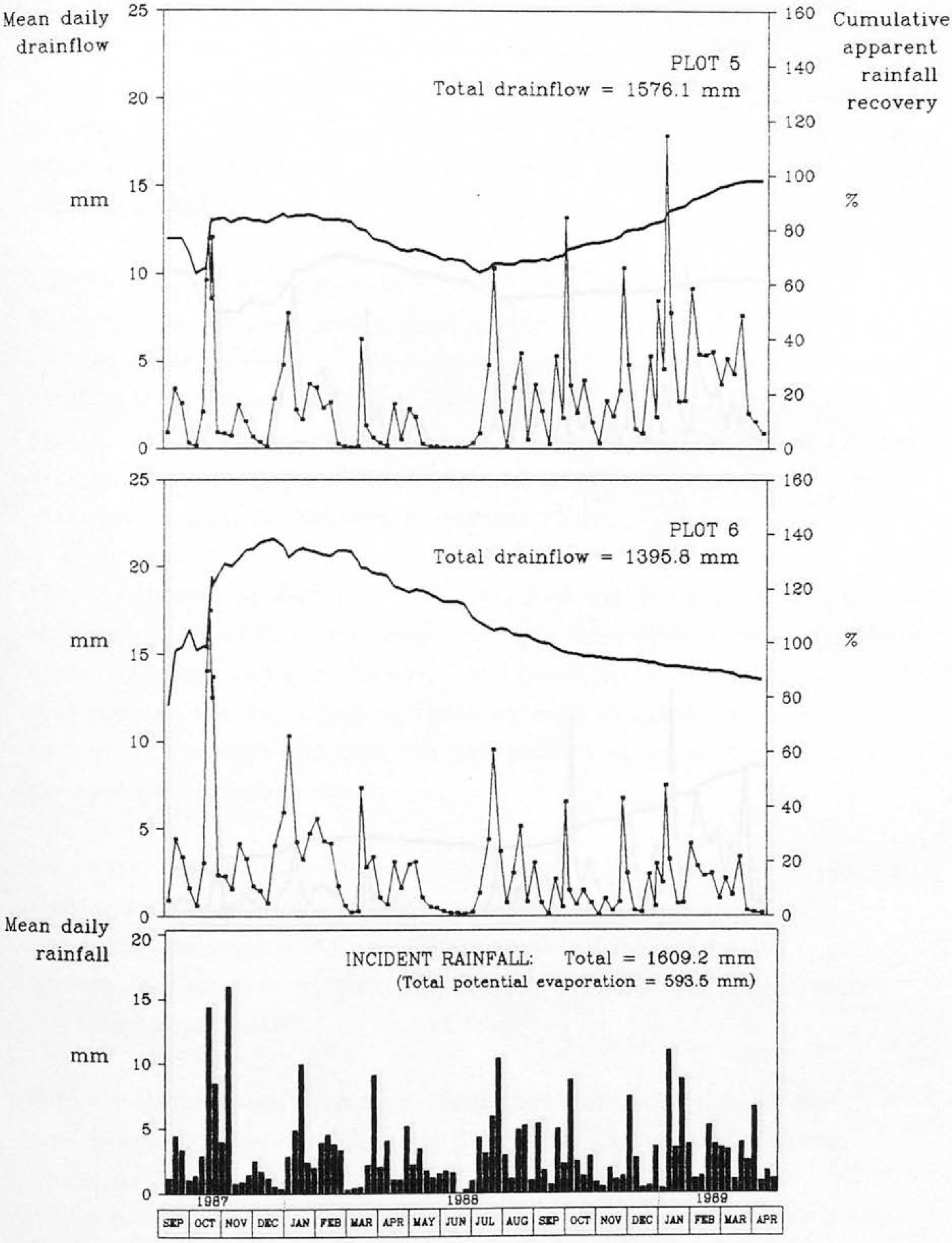
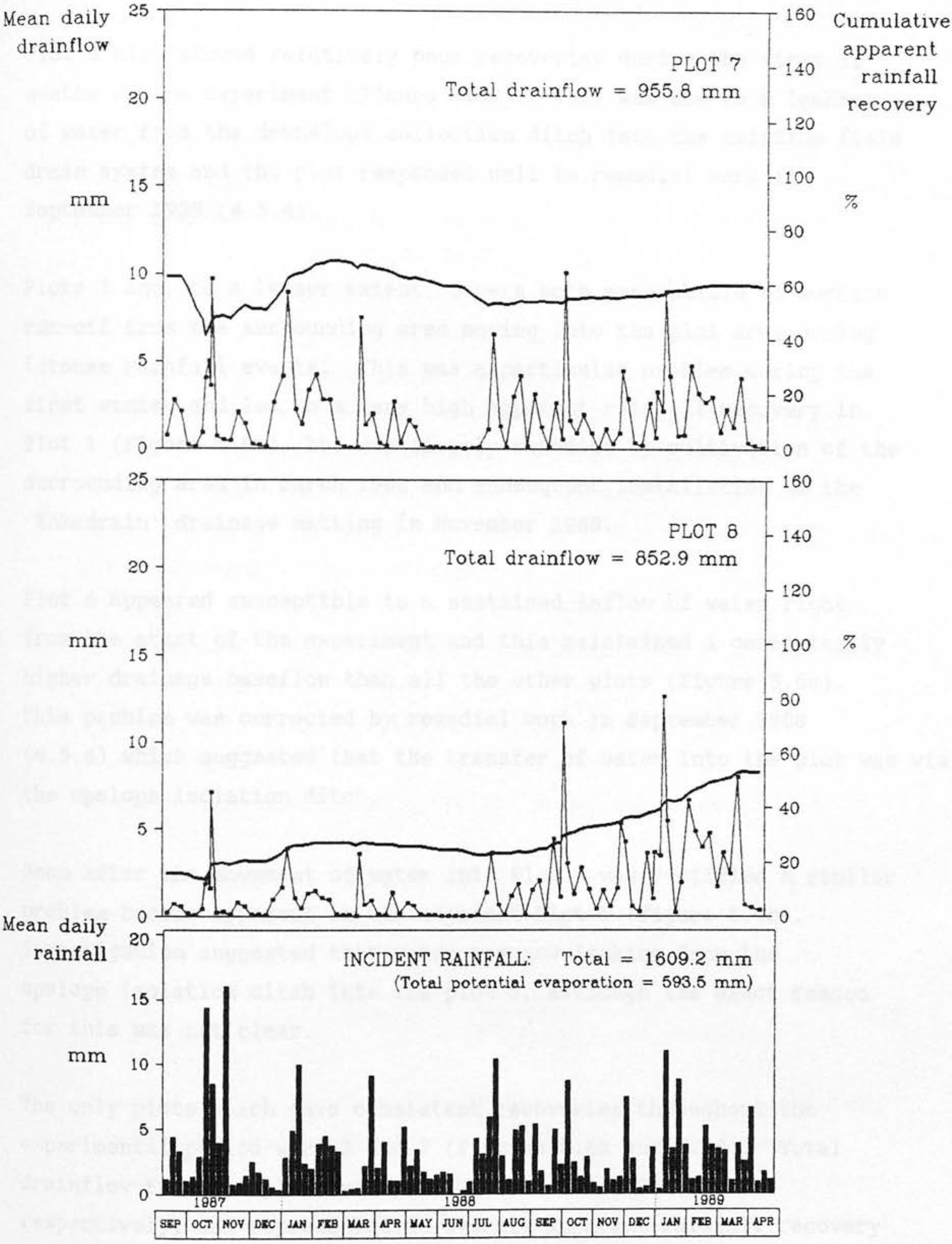


Figure 5.6d: Mean daily drainflow (mm) and cumulative % apparent recovery of incident rainfall from Plots 7 and 8 during the period 4 September, 1987 to 19, April 1989



rectify this failed and the final recovery of incident rainfall was only 4.0%. Reasons for this major loss proved difficult to determine at the time, but it was subsequently found that the pipe to the instrument pit was blocked (Vinten *et al.* 1991) and largely preventing the discharge of plot drainflow.

Plot 8 also showed relatively poor recoveries during the first 12 months of the experiment (Figure 5.6d). This was due to a leakage of water from the downslope collection ditch into the existing field drain system and the plot responded well to remedial work in September 1988 (4.5.4).

Plots 1 and, to a lesser extent, 5 were both susceptible to surface run-off from the surrounding area moving into the plot area during intense rainfall events. This was a particular problem during the first winter and led to a very high apparent rainfall recovery in Plot 1 (Figure 5.6a), but was largely remedied by cultivation of the surrounding area in March 1988 and subsequent installation of the 'Enkadrain' drainage matting in November 1988.

Plot 6 appeared susceptible to a sustained inflow of water right from the start of the experiment and this maintained a consistently higher drainage baseflow than all the other plots (Figure 5.6c). This problem was corrected by remedial work in September 1988 (4.5.4) which suggested that the transfer of water into the plot was via the upslope isolation ditch.

Soon after the movement of water into Plot 6 was rectified a similar problem became apparent in the adjacent Plot 5 (Figure 5.6c). Investigation suggested that water was now leaking from the upslope isolation ditch into the plot 5, although the exact reason for this was not clear.

The only plots which gave consistent recoveries throughout the experimental period were 2 and 7 (Figures 5.6a and 5.6d). Total drainflow from the plots was similar (965.4 and 955.8 mm respectively) and represented an overall apparent rainfall recovery of approximately 60%. This compared well with reported recoveries from other hydrologically isolated plots on heavy soil types, including the Boghall Pilot Plot (3.3.1). When further combined with potential evaporation data the recovery of total incident rainfall was approximately 96%, suggesting that hydrological isolation had the potential to be very successful on local glacial-till-derived soils.

Correction for variable drainflow recovery

Due to the problems with variable drainflow recovery from the plots it was necessary to make some correction to measurements of NO_3-N leaching losses. The following procedure was used for all plots :

i) Having identified Plots 2 and 7 as behaving 'normally' the 'expected' weekly drainflow (mm), D_{exp}, from all the plots was estimated by the mean weekly drainflow (mm) of plots 2 and 7. This mean value took some account of the possible differences in soil physical characteristics between plots and potential evapotranspiration losses between treatments during active crop growth (*i.e.* Plot 2 = zero N; Plot 7 = recommended N), but it still remained a major approximation;

ii) The corrected weekly drainflow (l), D_{corr}, for each plot was calculated as :

$$D_{corr} = D_{exp}A_{tot} \tag{21}$$

- where A_{tot} was the plot 'total' area (see below);

iii) The drainflow corrected weekly NO_3-N loading for each plot (g $plot^{-1}$), LD_{corr}, was calculated as :

$$LD_{corr} = \frac{D_{corr}[NO_3\text{-}N]}{1000} \tag{22}$$

Because of the low drainflow in Plot 3 there were considerably fewer water samples taken and so fewer NO_3-N concentration values. Therefore LD_{corr} was not calculated on a weekly basis, but rather over whatever time period was required to include a NO_3-N concentration value.

This correction procedure involved two assumptions :

A. *For plots with low rainfall recovery (notably Plot 3) the flow weighted mean NO_3-N concentration of the measured/sampled drainflow represented an unbiased estimate of the flow weighted mean NO_3-N concentration of the 'expected' drainflow.*

In other words, it was assumed that there was no significant bias in the estimation of the true mean NO_3-N concentration of plot drainage water arising from only sampling drainflow generated by relatively large plot discharges following heavy rainfall. This assumption was tested using data from spot sampled drainflow collected from Plot 5 on an 8-hour cycle between 14 December, 1987 and 2 February, 1988 using the automatic liquid samplers (3.1.5).

Estimation of the flow-weighted average NO_3-N concentration for different drainflow ranges revealed no significant bias when only samples from high drainflows were used (Table 5.8). Since there was good agreement in the estimation of flow-weighted NO_3-N concentrations when using the automatic liquid samplers and the simple flow-dividing device (Vinten *et al.* 1991), it seems reasonable to assume that no bias arose when the flow-dividing device effectively only sampled from high plot discharges. A possible exception is where there was surface applied fertiliser N present, since this may be more readily leached at high flow

Table 5.8: Estimated flow-weighted average NO_3-N concentrations for spot samples from different drainflow ranges in Plot 5 (14 December, 1987 - 2 February, 1988)

Minimum flow rate sampled (mm hr^{-1})	% of time flow exceeded	Average $[NO_3\text{-}N]$ (mg l^{-1})
0	100	1.34
0.082	35	1.33
0.283	12	1.29

rates due to the occurrence of 'bypass' flow in the large, rapidly draining macropores found in this heavy, structured soil type (Vinten and Redman 1990).

B. *For plots with high rainfall recovery the water moving into a plot contained the same NO_3-N concentration as water derived from the plot.*

This was probably a reasonable assumption for any ground/drain water movement into the plots, given the small range of NO_3-N concentrations from the different treatments during most of the experimental period (with the exception of Plots 3 and 8 in the autumn 1987). It would, however, have tended to mask any treatment differences which occurred. Also, if surface run-off was contributing significantly towards high rainfall recoveries, this correction procedure will have underestimated NO_3-N losses because the NO_3-N content of surface water will have been negligible. Unfortunately it was not possible to identify the relative contribution of surface, ground and drain water movement into the plots.

Correction for recovery from 'non-plot' area

There were effectively 3 areas associated with the investigation of NO_3-N losses from the hydrologically isolated plots:

a) *'agronomic' area,* A_{agr}

- plot area to which N treatments applied (mean = 309.4 m²);

b) *'hydrological' area,* A_{hyd}

- 'agronomic' area plus 50% of the adjacent guard areas assumed to be contributing water to the plot drainage collection system (mean = 331.7 m²);

c) *'total' area,* A_{tot}

- 'hydrological' area plus the area of downslope collection ditch and exposed tipping buckets receiving incident rainfall (mean = 337.6 m²).

Because the guard areas contributed some NO_3-N to the measured leaching losses it was considered necessary to further correct leaching losses on the basis of area. The following procedure was used for plots 1,3,4,6,7 and 8, assuming that the NO_3-N contribution from the guard areas could be estimated from the NO_3-N losses from the zero N plots (2 and 5) :

i) The weekly NO_3-N loading per unit area from the guard areas ($g\ m^{-2}$), GA, was estimated by :

$$GA = \frac{LD_{corr}\ (\text{Plot } 2) + LD_{corr}\ (\text{Plot } 5)}{A_{hyd}\ (\text{Plot } 2) + A_{hyd}\ (\text{Plot } 5)} \tag{23}$$

ii) The area corrected weekly NO_3-N loading ($g\ plot^{-1}$) from plots 1,3,4,6,7 and 8, LA_{corr}, was calculated as :

$$LA_{corr} = LD_{corr} - \left[GA(A_{hyd} - A_{agr})\right] \tag{24}$$

iii) The final corrected weekly NO_3-N loading ($kg\ ha^{-1}$) from plots 1,3,4,6,7 and 8, L_{corr}, was calculated as :

$$L_{corr} = LA_{corr}\ \frac{10}{A_{agr}} \tag{25}$$

- whilst L_{corr} for plots 2 and 5 was calculated as :

$$L_{corr} = LA_{corr}\ \frac{10}{A_{hyd}} \tag{26}$$

The potential contribution of N input as incident rainfall on the downslope plot collection ditches and tipping buckets was ignored.

5.3.2 LEACHING LOSSES

Figures 5.5a - 5.5d include the drainflow NO_3-N concentrations ($mg\ l^{-1}$), uncorrected and corrected cumulative NO_3-N leaching losses ($kg\ ha^{-1}$) for all the plots during the period 4 September 1987 to 19 April 1989.

Figure 5.7a: NO_3-N concentrations (mg l^{-1}) and uncorrected/corrected NO_3-N leaching losses (kg ha^{-1}) from Plots 1 and 2 during the period 4 September, 1987 to 19, April 1989

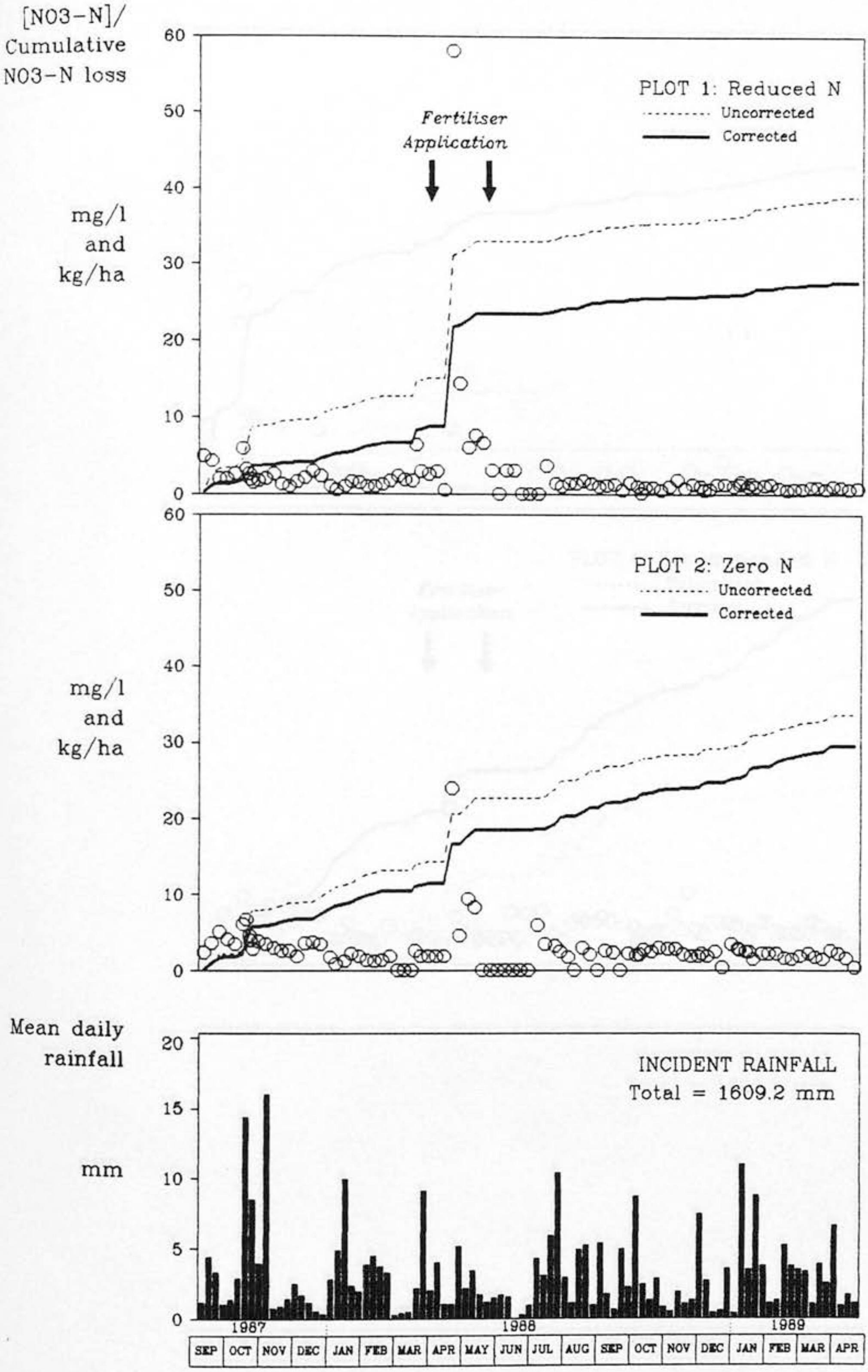


Figure 5.7b: NO_3-N concentrations (mg l^{-1}) and uncorrected/corrected NO_3-N leaching losses (kg ha^{-1}) from Plots 3 and 4 during the period 4 September, 1987 to 19, April 1989

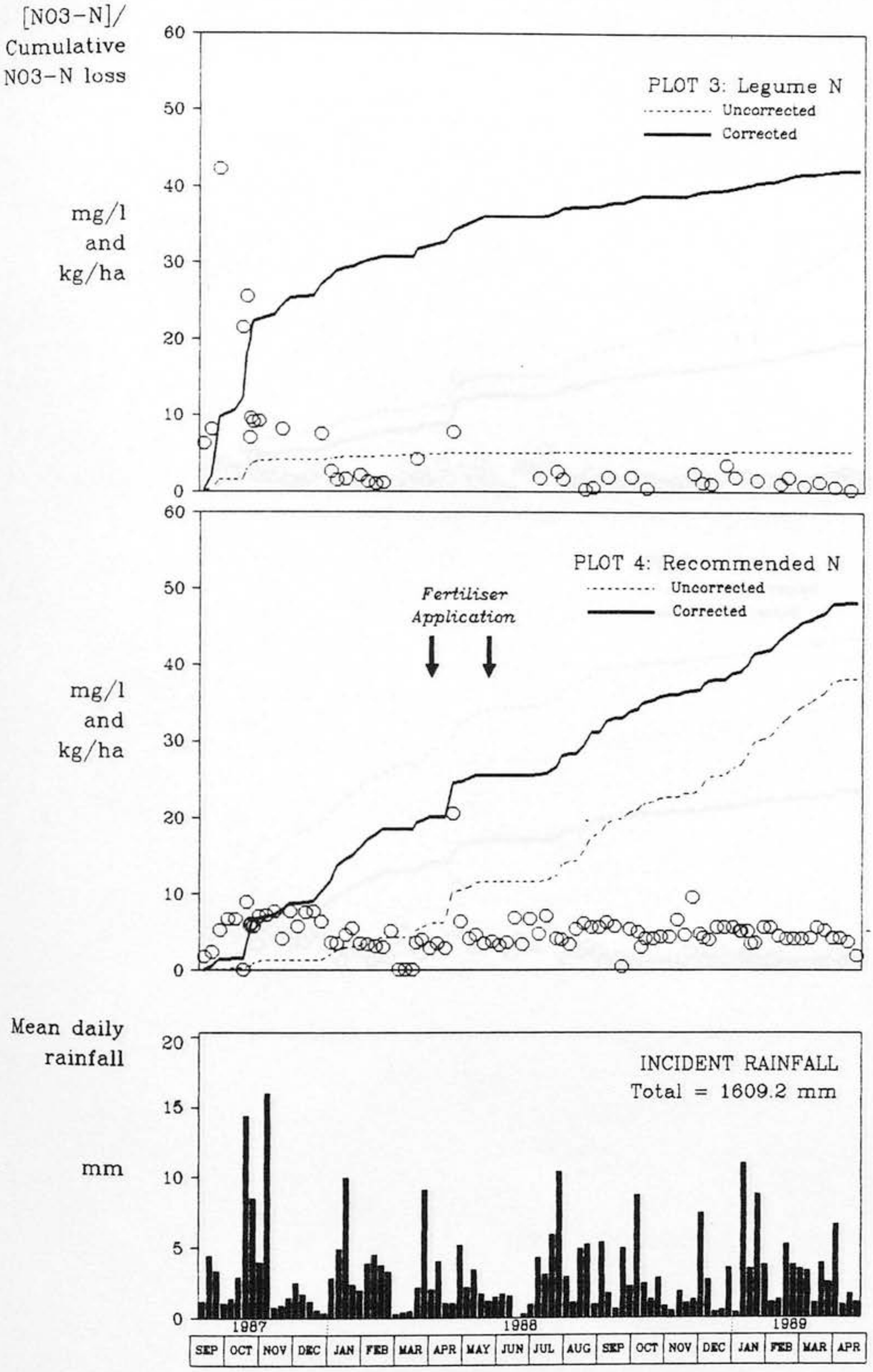


Figure 5.7c: NO_3-N concentrations (mg l^{-1}) and uncorrected/corrected NO_3-N leaching losses (kg ha^{-1}) from Plots 5 and 6 during the period 4 September, 1987 to 19, April 1989

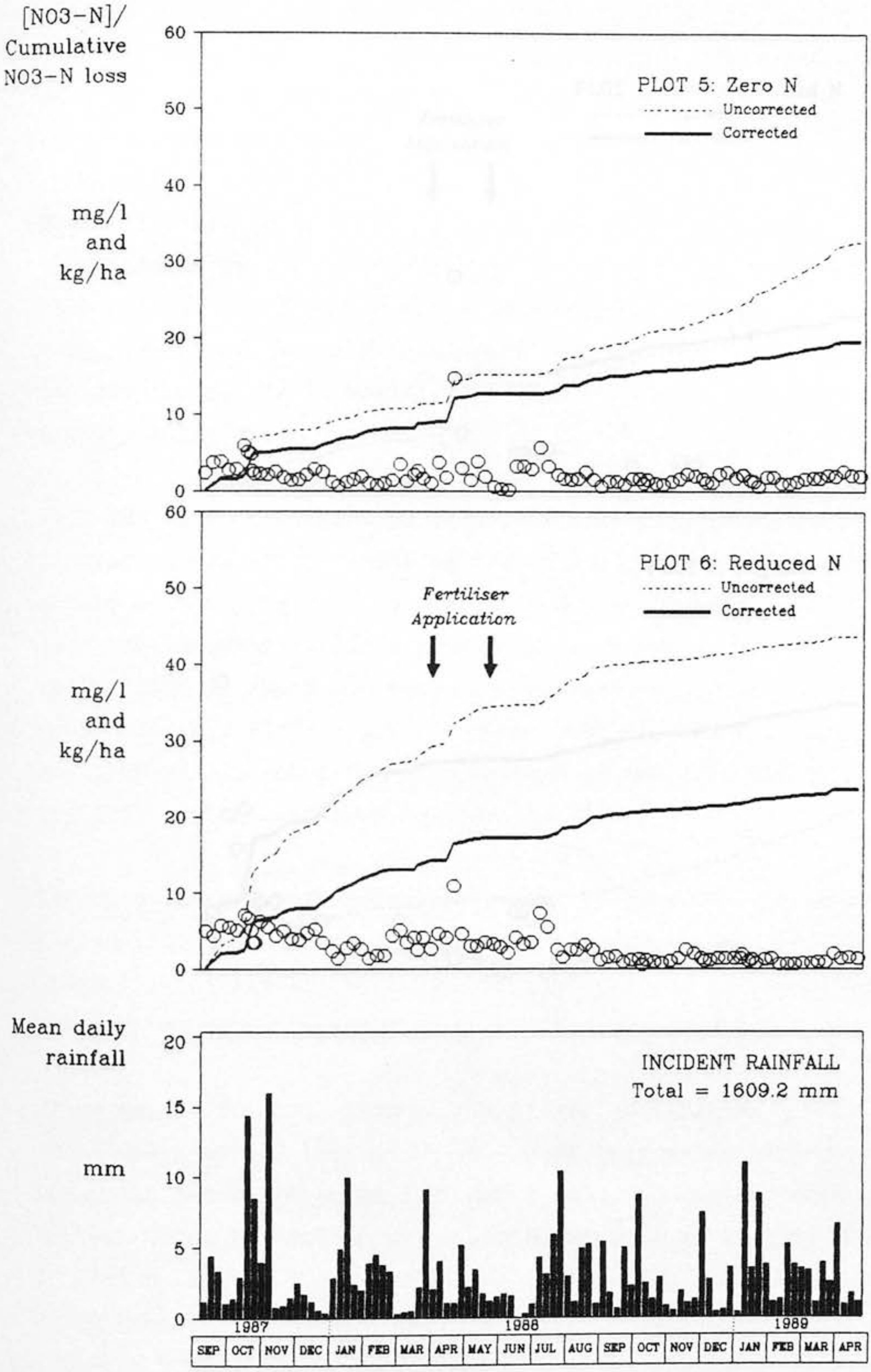
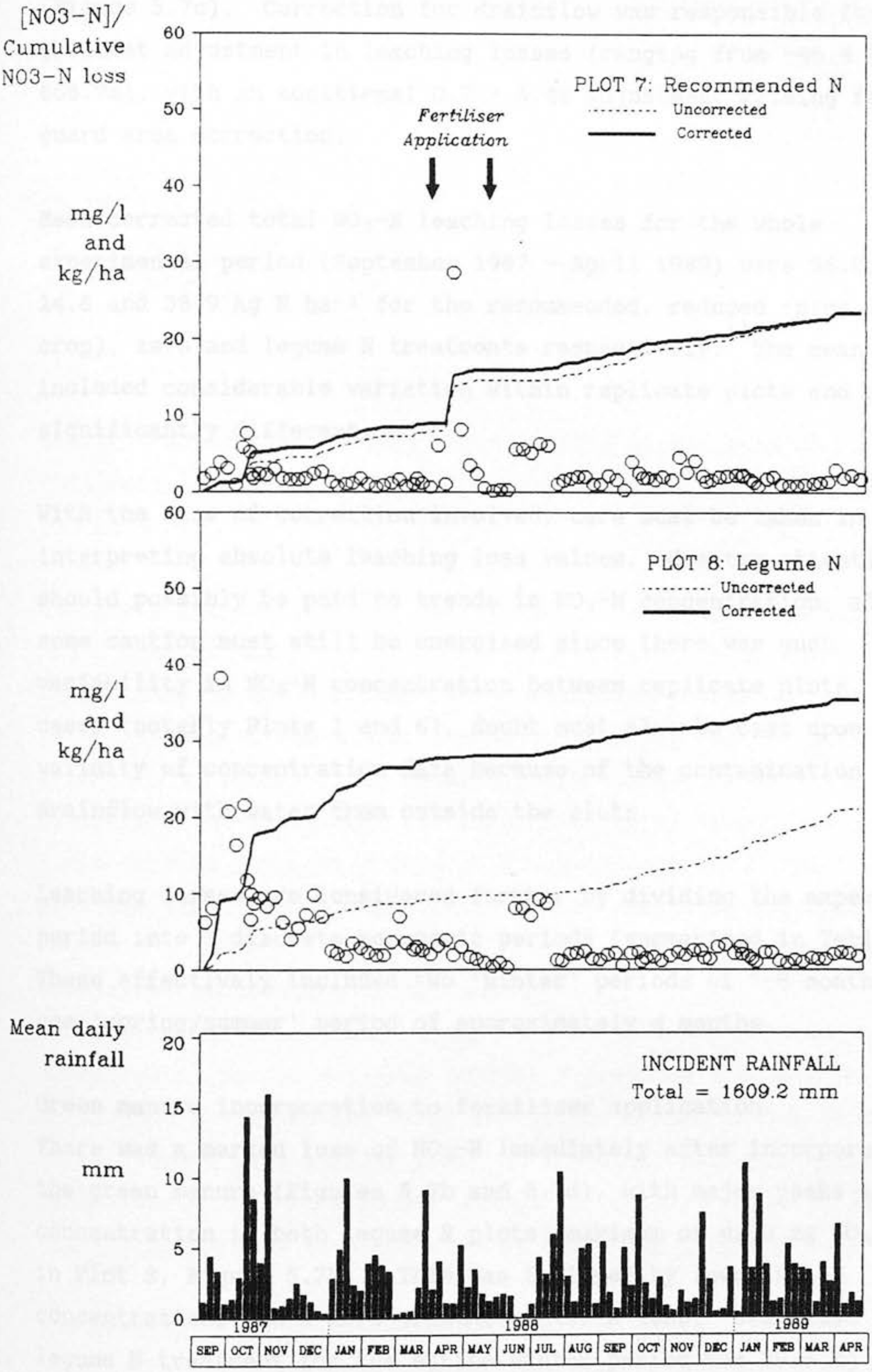


Figure 5.7d: NO_3-N concentrations (mg l^{-1}) and uncorrected/corrected NO_3-N leaching losses (kg ha^{-1}) from Plots 7 and 8 during the period 4 September, 1987 to 19, April 1989



In many plots, the corrected leaching losses were very different from the uncorrected losses. Maximum *positive* correction was the addition of 36.9 kg N ha^{-1} in Plot 3 (Figure 5.7b) and maximum *negative* correction the subtraction of 20.1 kg N ha^{-1} in Plot 6 (Figure 5.7c). Correction for drainflow was responsible for the greatest adjustment in leaching losses (ranging from -46.4 to 668.7%), with an additional 0.7 - 5.4% adjustment arising from guard area correction.

Mean corrected total NO_3-N leaching losses for the whole experimental period (September 1987 - April 1989) were 36.0, 25.8, 24.6 and 38.9 kg N ha^{-1} for the recommended, reduced (plus cover crop), zero and legume N treatments respectively. The mean values included considerable variation within replicate plots and were not significantly different.

With the size of correction involved, care must be taken in interpreting absolute leaching loss values. Greater attention should possibly be paid to trends in NO_3-N concentration, although some caution must still be exercised since there was much variability in NO_3-N concentration between replicate plots. In some cases (notably Plots 1 and 6), doubt must also be cast upon the validity of concentration data because of the contamination of drainflow with water from outside the plots.

Leaching losses are considered further by dividing the experimental period into 3 discrete agronomic periods (summarised in Table 5.9). These effectively included two 'winter' periods of 7-8 months and one 'spring/summer' period of approximately 4 months.

Green manure incorporation to fertiliser application

There was a marked loss of NO_3-N immediately after incorporation of the green manure (Figures 5.7b and 5.7d), with major peaks in NO_3-N concentration in both legume N plots (maximum of 42.2 mg NO_3-N l^{-1} in Plot 3, Figure 5.7b). This was followed by lower NO_3-N concentrations and a more gradual winter N loss. Mean loss from the legume N treatment for the entire winter period was 29.5 kg N ha^{-1} (Figure 5.9), of which over 68% occurred before the end of October

Table 5.9: Summary of the main agronomic periods, including total rainfall, potential evaporation and 'expected' drainflow, D_{exp} (mm)

Period:	Dates:	Crop:	Rainfall:	Pot. evap.:	D_{exp}:
Green manure incorporation to fertiliser applic.	4/9/87 - 29/3/88	Winter barley	590.2	115.0	421.8
Fertiliser applic. to harvest	30/3/88 - 26/7/88	Winter barley	312.3	243.7	124.0
Harvest to following crop	27/7/88 - 20/4/89	Stubble fallow/ winter rye	706.7	234.8	414.8

1987. Leaching losses were significantly higher from the legume N treatment than from the spring barley that had previously received 120 kg N ha^{-1}.

The plots following spring barley did not show the same marked NO_3-N loss as the legume N plots (Figure 5.9). Although there were small peaks in NO_3-N concentration following ploughing and establishment of the winter barley and a rapid loss of 2.4 - 3.8 kg N ha^{-1} (*e.g.* Figure 5.7c) between 13-19 October, 1987 coinciding with the first heavy rain (89 mm in 6 days) after cultivation. There were no apparent trends in NO_3-N concentration related to previous fertiliser N application to the spring barley crop.

Fertiliser application to harvest

The highest NO_3-N concentration (58.0 mg l^{-1}) of the whole experimental period occurred in Plot 1 (reduced N treatment) during April 1988 following the first fertiliser N application (Figure 5.7a). There were similar peaks in the other plots receiving fertiliser N, and to a lesser extent in the zero N plots, and these all led to a rapid loss of NO_3-N at this time (Figures 5.7a - 5.7d).

The greatest weekly loss event was between 12 and 19 April 1988 when 36.1 mm rainfall led to the mean loss of 4.1, 7.6 and 5.4 kg N ha^{-1}

Figure 5.8: Flow-weighted mean NO_3-N drainflow concentrations (mg l^{-1}) for the three periods following incorporation of the green manure, fertiliser application and harvest

Plot numbers are in histograms and treatment means in parentheses

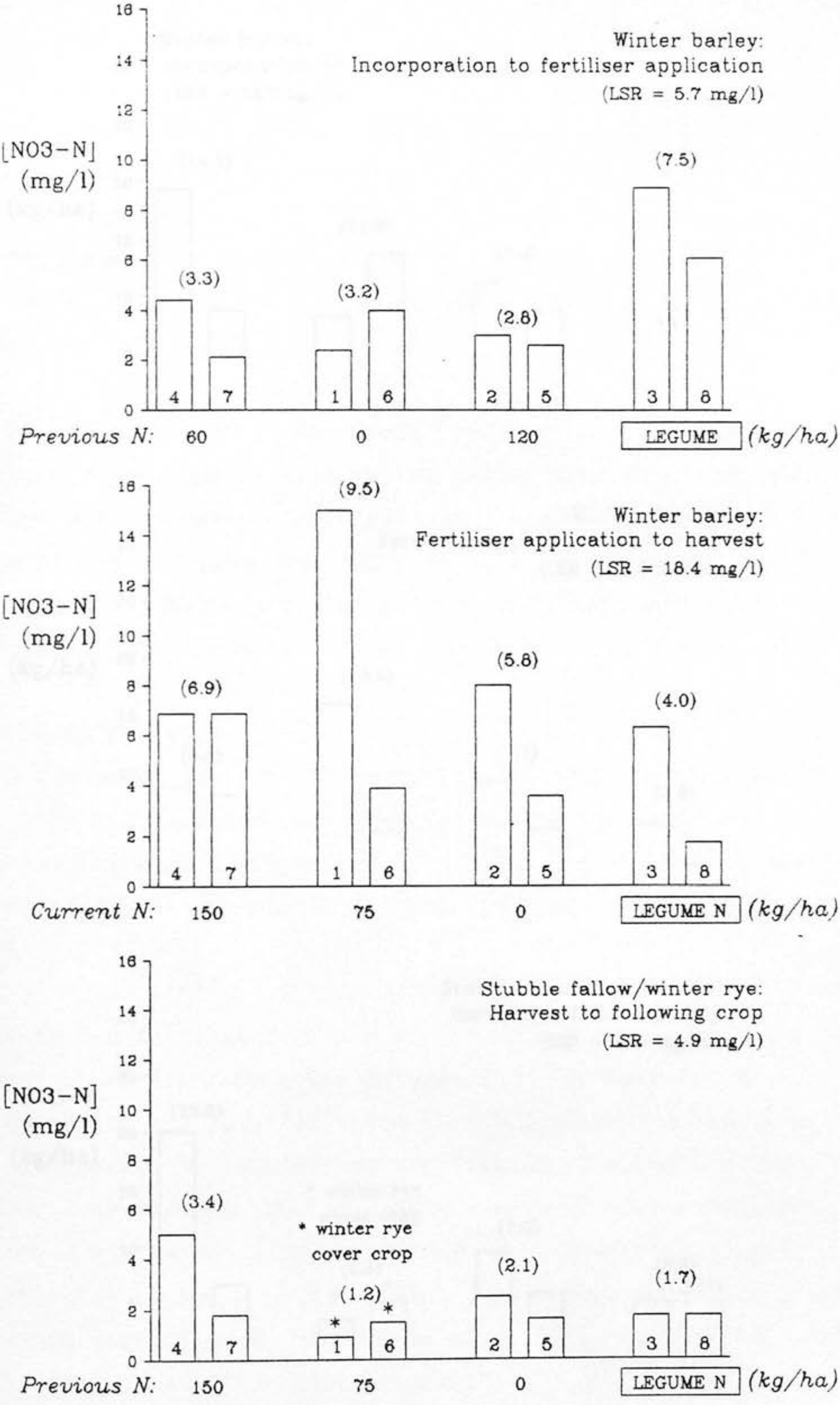
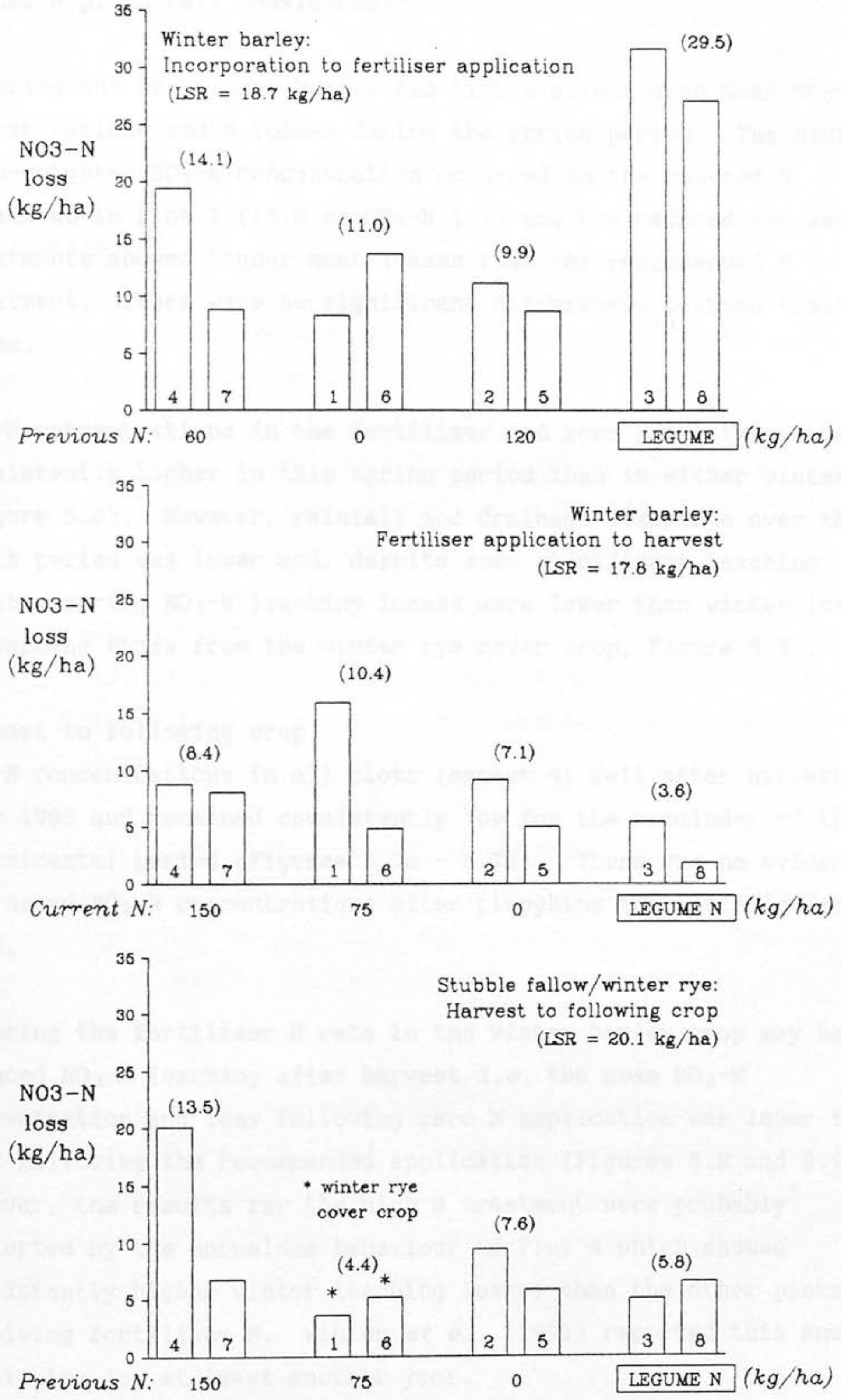


Figure 5.9: Estimated NO_3-N leaching losses (kg N ha^{-1}) during the three periods following incorporation of the green manure, fertiliser application and harvest

Plot numbers are in histograms and treatment means in parentheses



from the zero, reduced and recommended N plots respectively. During this same week, only 0.9 kg N ha^{-1} was lost from the legume N plots and, whereas the flow-weighted mean NO_3-N concentrations of all other plots rose in the spring period, mean concentrations from the legume N plots fell considerably.

Reducing the fertiliser N rate had little effect upon mean NO_3-N concentrations and N losses during the spring period. The highest flow-weighted NO_3-N concentration occurred in the reduced N treatment in Plot 1 (15.0 mg NO_3-N l^{-1}) and the reduced and zero N treatments showed higher mean losses than the recommended N treatment. There were no significant differences between treatment means.

NO_3-N concentrations in the fertiliser and zero N treatments were consistently higher in this spring period than in either winter (Figure 5.8). However, rainfall and drainage discharge over the 4 month period was lower and, despite some significant leaching events, spring NO_3-N leaching losses were lower than winter losses (excepting those from the winter rye cover crop, Figure 5.9).

Harvest to following crop

NO_3-N concentrations in all plots (except 4) fell after harvest in July 1988 and remained consistently low for the remainder of the experimental period (Figures 5.7a - 5.7d). There was no evidence of increased NO_3-N concentrations after ploughing the plots in March 1989.

Reducing the fertiliser N rate in the winter barley crop may have reduced NO_3-N leaching after harvest *i.e.* the mean NO_3-N concentration and loss following zero N application was lower than that following the recommended application (Figures 5.8 and 5.9). However, the results for the high N treatment were probably distorted by the anomalous behaviour of Plot 4 which showed consistently higher winter leaching losses than the other plots receiving fertiliser N. Vinten *et al.* (1991) reported this anomaly continuing for at least another year.

Flow-weighted NO_3-N concentrations and losses in all plots (except 4) were lower than in the previous 1987/88 winter, even though they were predominantly under a stubble fallow rather than a growing crop (Figures 5.8 and 5.9). Establishment of the rye cover crop in the reduced N plots in August 1988 appeared to further reduce NO_3-N concentrations and losses, but there were no significant differences between treatment means. The mean NO_3-N concentration and loss from the legume N treatment remained below that of the zero N treatment and very much lower than levels in the previous winter.

^{15}N leaching data

Direct measurement of leaching losses from the ^{15}N-labelled legume material incorporated in Plot 3 (4.5.3) was abandoned in December 1988 for two reasons:

a) the very low drainflow recovery of the plot (5.3.1) was making it difficult to regularly collect water samples of sufficient volume (≈500 ml) for ^{15}N analysis;

b) the high levels of N_2 fixation in the peas (5.2.3) had diluted the ^{15}N content of the legume material to such an extent that it was close to atmospheric enrichment (≈0.03 atom% excess).

The ^{15}N enrichment of the drainage water from Plot 7 was monitored from application of the first fertiliser N dressing on 29 March 1988 until 25 May 1988, shortly after the second dressing. Excess enrichments in samples analysed after this date were found to be negligible.

Total NO_3-N leaching loss from Plot 7 during this period was 7.1 kg ha^{-1}, of which 5.7 kg ha^{-1} was derived from the labelled-fertiliser N (7.6% of application). 6.3 kg ha^{-1} was lost from the plot during the heavy rainfall between 12 and 19 April (Figure 5.7d) and included 5.4 kg ha^{-1} (86%) of fertiliser N.

5.3.3 RAINFALL N INPUTS

The mean weighted NO_3-N and NH_4-N concentrations in rainfall between October 4, 1988 and June 27, 1989 were 0.4 and 0.9 mg l^{-1} respectively, with concentrations ranging from 0.1 - 4.8 mg NO_3-N l^{-1} and 0.5 - 2.7 mg NH_4-N l^{-1}.

The rainfall N input between October 1988 and June 1989 was therefore assumed to be equivalent to 0.4 kg NO_3-N and 0.9 kg NH_4-N ha^{-1} 100 mm^{-1} rainfall. When extrapolated to the whole experimental period (total rainfall = 1609.2 mm), the total N input was approximately 20.9 kg N ha^{-1} *i.e.* 6.4 kg NO_3-N and 14.5 kg NH_4-N ha^{-1}.

Estimated rainfall N inputs for the main agronomic periods (5.3.2) are summarised in Table 5.10.

5.3.4 CROP N UPTAKE

Winter barley

The winter barley yield data at harvest is summarised in Table 5.11. Dry matter yield, crop N uptake and grain yield was highest in the recommended N treatment and declined linearly with decreasing fertiliser N application. Yield data for the legume N treatment lay between the zero and reduced N treatments. All treatment means were significantly different.

Table 5.10: Summary of estimated rainfall N inputs (kg N ha^{-1}) during main agronomic periods

Period:	NO_3-N:	NH_4-N:	Total N:
Green manure incorporation to fertiliser applic.	2.4	5.3	7.7
Fertiliser applic. to harvest	1.2	2.9	4.1
Harvest to following crop	2.8	6.3	9.1
Total:	6.4	14.5	20.9

Prior to fertiliser application in spring 1988, crop N uptake was significantly higher in the legume N treatment (Figure 5.10). There was no relationship between fertiliser N application to the preceding spring barley and N uptake by the winter barley during the autumn/winter.

N uptake in the recommended and reduced N treatments increased after fertiliser application (Figure 5.10) and greatly exceeded that from the legume N treatment. Most of the increased uptake was derived from fertiliser N. Although uptake in the zero N treatment continued to be lowest of all treatments, it was not significantly different from the legume N treatment during the spring period.

The pattern of soil and fertiliser N uptake by the winter barley was similar to that found in the spring barley (5.2.2). Decreasing the fertiliser N application significantly reduced the uptake of fertiliser N (Table 5.12) and increased the proportion of crop N derived from the soil (Table 5.13). There was also a linear decline

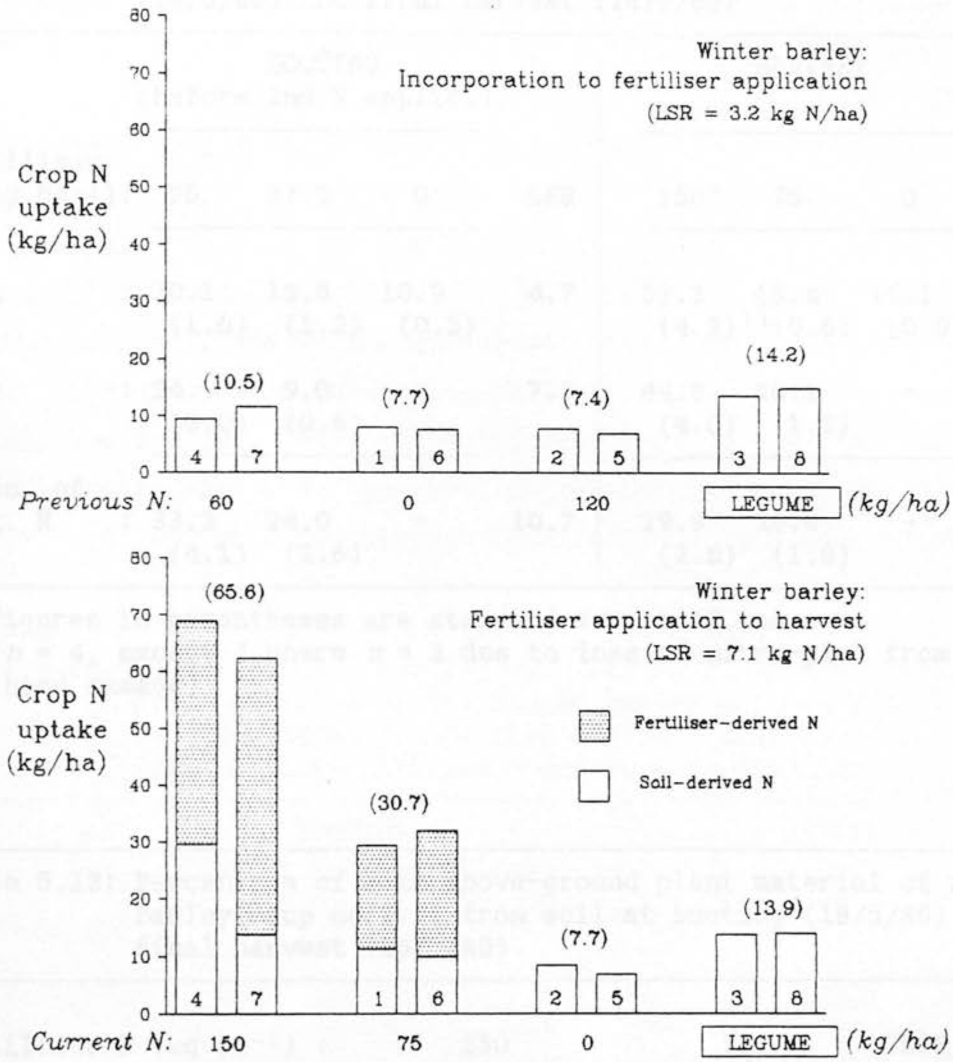
Table 5.11: Mean yield data for winter barley, September 1987 - July 1988

		Treatment :				
		Recomm. N	Reduced N	Zero N	Legume N	LSR
N rate (kg ha^{-1})	:	150	75	0	358.2	
Plots	:	4 + 7	1 + 6	2 + 5	3 + 8	
Total Dry Matter Yield* (t ha^{-1})	:	9.55 (0.21)	6.16 (0.08)	2.14 (0.19)	4.17 (0.08)	1.04
Total N Uptake (kg N ha^{-1})	:	76.1 (2.0)	38.5 (0.9)	15.1 (0.9)	28.2 (0.4)	5.0
Grain Yield at 15% moisture content (t ha^{-1})	:	4.71 (0.12)	2.79 (0.21)	0.84 (0.08)	1.87 (0.04)	0.54

() Figures in parentheses are standard errors of means (n = 4, except * where n = 2)

Figure 5.10: Crop N uptake (kg N ha^{-1}) during the two agronomic periods following incorporation of the green manure and fertiliser application

Plot numbers are in histograms and treatment means in parentheses



in absolute soil N uptake as fertiliser application was decreased, with uptake in the reduced and zero N treatments significantly lower than the recommended N treatment (Table 5.12). These effects were seen at booting (*i.e.* after a single split dressing) and harvest (*i.e.* after both dressings). In both fertiliser N treatments the proportion of crop N derived from the soil declined between booting and harvest (Table 5.13).

Table 5.12: Uptake of soil and fertiliser N (kg ha^{-1}) in above-ground plant material of winter barley crop at booting (18/5/88) and final harvest (26/7/88)

	BOOTING (before 2nd N applic.)				HARVEST			
Fertiliser N (kg ha^{-1}):	75	37.5	0	**LSR**	150*	75	0	**LSR**
N_{dfs} :	20.1 (1.6)	13.8 (1.2)	10.9 (0.5)	**4.7**	31.3 (4.2)	18.4 (0.6)	15.1 (0.9)	**8.2**
N_{dff} :	24.9 (3.0)	9.0 (0.6)	-	**7.5**	44.8 (4.0)	20.1 (1.3)	-	**9.4**
% Rec. of Fert. N :	33.2 (4.1)	24.0 (1.6)	-	**10.7**	29.9 (2.6)	26.8 (1.8)	-	**7.8**

() Figures in parentheses are standard errors of means (n = 4, except * where n = 3 due to loss of micro-plot from bird damage)

Table 5.13: Percentage of N in above-ground plant material of winter barley crop derived from soil at booting (18/5/88) and final harvest (26/7/88)

Fertiliser N (kg ha^{-1}) :	150	75	LSR
Shoot at booting :	44.7 (3.3)	60.5 (2.5)	**10.1**
Grain :	40.0 (5.3)	46.6 (2.2)	**13.2**
Straw :	43.3 (5.8)	49.7 (2.5)	**14.5**

() Figures in parentheses are standard errors of means (n = 4)

The % recovery of applied ^{15}N-labelled fertiliser in the crop was within the range of 24-34% for both fertiliser N treatments and sample dates (Table 5.12).

There was a reduction in N uptake from the legume N treatment between March and April 1988. It is unlikely that this was an actual loss of crop N, but it probably reflected at least some reduction in the rate of uptake. At harvest the 'apparent' uptake of legume-derived N (calculated by difference) was 13.1 kg N ha^{-1}, a recovery of only 4.2% of the total legume N incorporated into the soil (5.2.3). Despite the low enrichment of the ^{15}N-labelled green manure material in Plot 3 (5.3.2), detectable increases in ^{15}N enrichment were found in the subsequent barley crop and were tentatively used to calculate legume-derived N uptake (N_{dfl}) at booting and harvest. N_{dfl} estimated from isotope dilution was approximately twice the 'apparent' uptake calculated by difference (Table 5.14) and at final harvest equalled 22.7 kg N ha^{-1} (7.3% of

Table 5.14: Uptake of soil and legume N (kg ha^{-1}) in above-ground plant material of the winter barley crop in Plot 3 at booting (18/5/88) and final harvest (26/7/88) as determined by difference and isotope dilution

	BOOTING (before 2nd N applic.)		HARVEST	
Incorporated legume N (kg ha^{-1}) :	310.2		310.2	
Method :	difference	isotopic*	difference	isotopic*
N_{dfs} :	10.9+ (0.5)	3.8 (0.4)	15.1+ (0.9)	5.5 (1.5)
N_{dfl} :	4.5 (1.2)	11.6 (0.4)	13.1 (1.3)	22.7 (1.3)
% Rec. of incorporated Legume N :	1.5 (0.3)	3.7 (0.1)	4.2 (0.4)	7.3 (0.4)

() Figures in parentheses are standard errors of means (n = 4, except * where n = 2)

+ zero N uptake assumed to be valid estimate of N_{dfs} for calculation of N_{dfl} by difference the incorporated legume N input). Accordingly the uptake of soil-derived N was found to be less in the legume N treatment than in the zero N.

Winter rye

Mean N uptake in the winter cover crop was 8.2 and 7.7 kg N ha^{-1} on 15 November, 1988 and 13 March, 1989 respectively (standard errors were 0.7 and 0.5 kg N ha^{-1}). The successive treatment means were not significantly different (LSR = 2.0).

There had been no weed control measures on the plots since herbicide application in autumn 1987 (4.4) and a number of annual weed species (*e.g.* groundsel, *Senecio vulgaris*) flourished in the stubble fallow after harvest. There was not time to sample all the plots, but weed samples were collected and analysed (as winter rye in 3.6.3) from Plots 1 and 4.

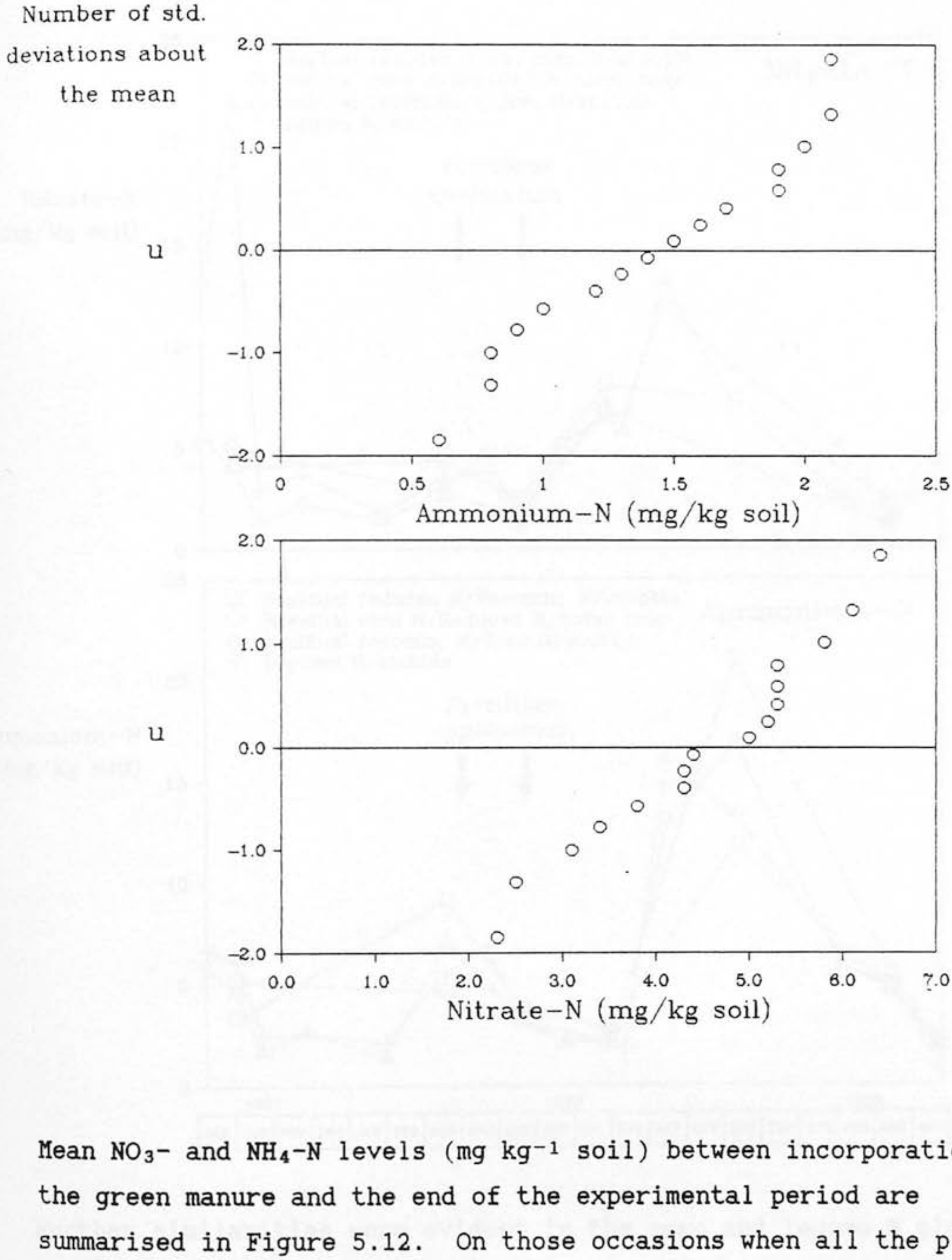
On 15 November, N uptake by the weeds was estimated to range between 5.0 and 20.4 kg N ha^{-1} *i.e.* in Plots 1 and 4 respectively. By 13 March the weeds had completely died back.

5.3.5 SOIL MINERAL N

Several authors (*e.g.* Macduff and White 1984) have reported a log-normal spatial distribution of NH_4-N and NO_3-N in soil samples and have used geometric, rather than arithmetic, means as an estimate of central tendency. The distribution of mineral N levels in soil samples from the Glencorse site was investigated at the end of the experimental period. On 20 April 1989, sixteen 20 cm soil cores were taken from Plot 2 in a 4x4 grid before the plot was drilled (the plot had received no fertiliser N since May 1987).

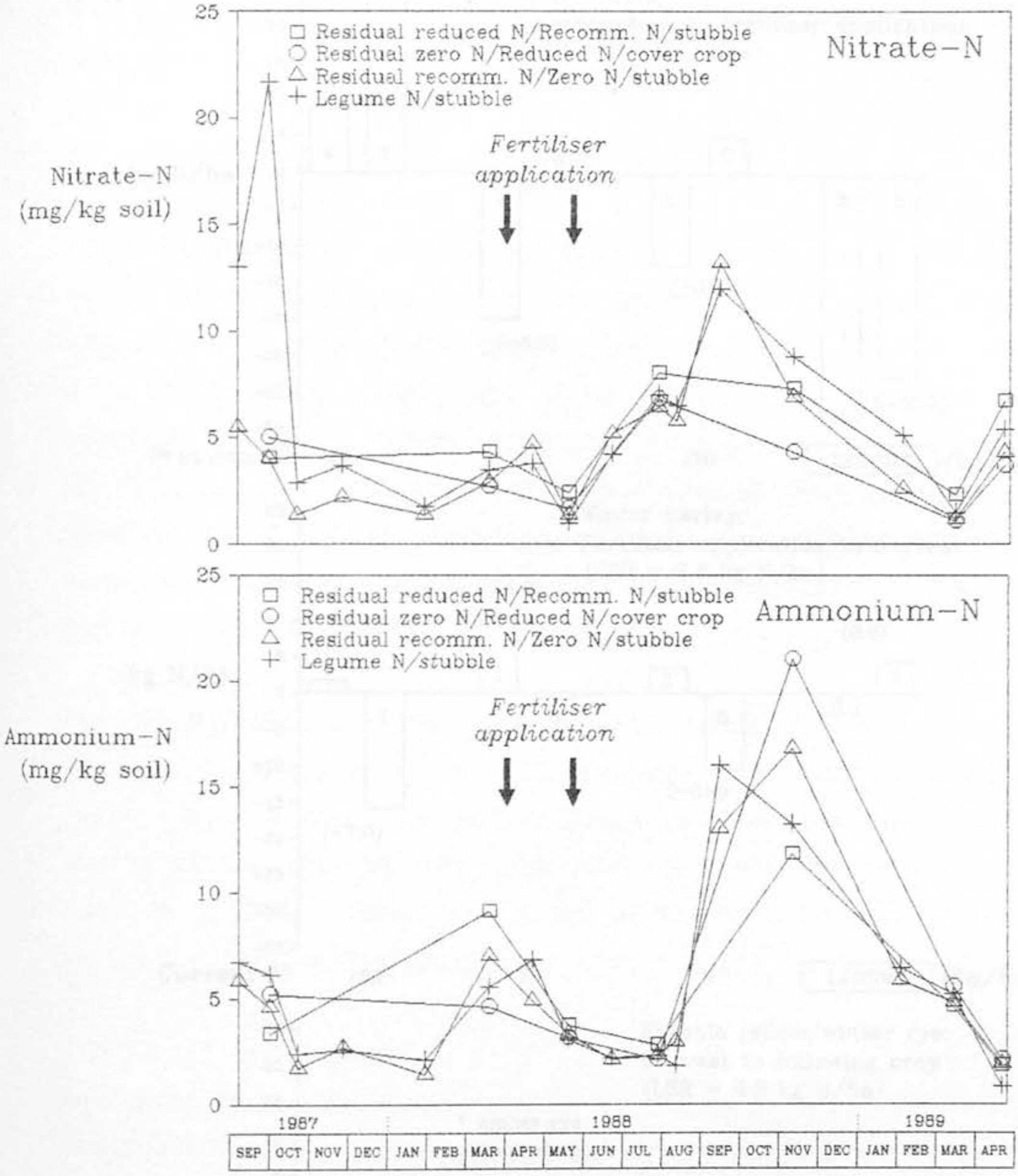
Using the limited mineral N data obtained simple fractile diagrams (Warrick and Nielsen 1980) were compiled (Figure 5.11). No evidence of a skewed distribution was found, and it was assumed that mineral N was normally distributed. All soil mineral N data were therefore prepared as arithmetic means (n = 4) for each treatment and sampling date.

Figure 5.11: Simple fractile diagrams of NO_3-N and NH_4-N (mg kg^{-1} soil) in 0-20 cm layer sampled on 20 April, 1989, where u = number of standard deviations about the mean



Mean NO_3- and NH_4-N levels (mg kg^{-1} soil) between incorporation of the green manure and the end of the experimental period are summarised in Figure 5.12. On those occasions when all the plots were sampled, the absolute level and seasonal fluctuation of mineral N was found to be similar in all four treatments. Spatial variability was high (CV = 21-83%) and some replicate plots showed marked variation in the temporal changes in soil mineral N content (Figure 5.13).

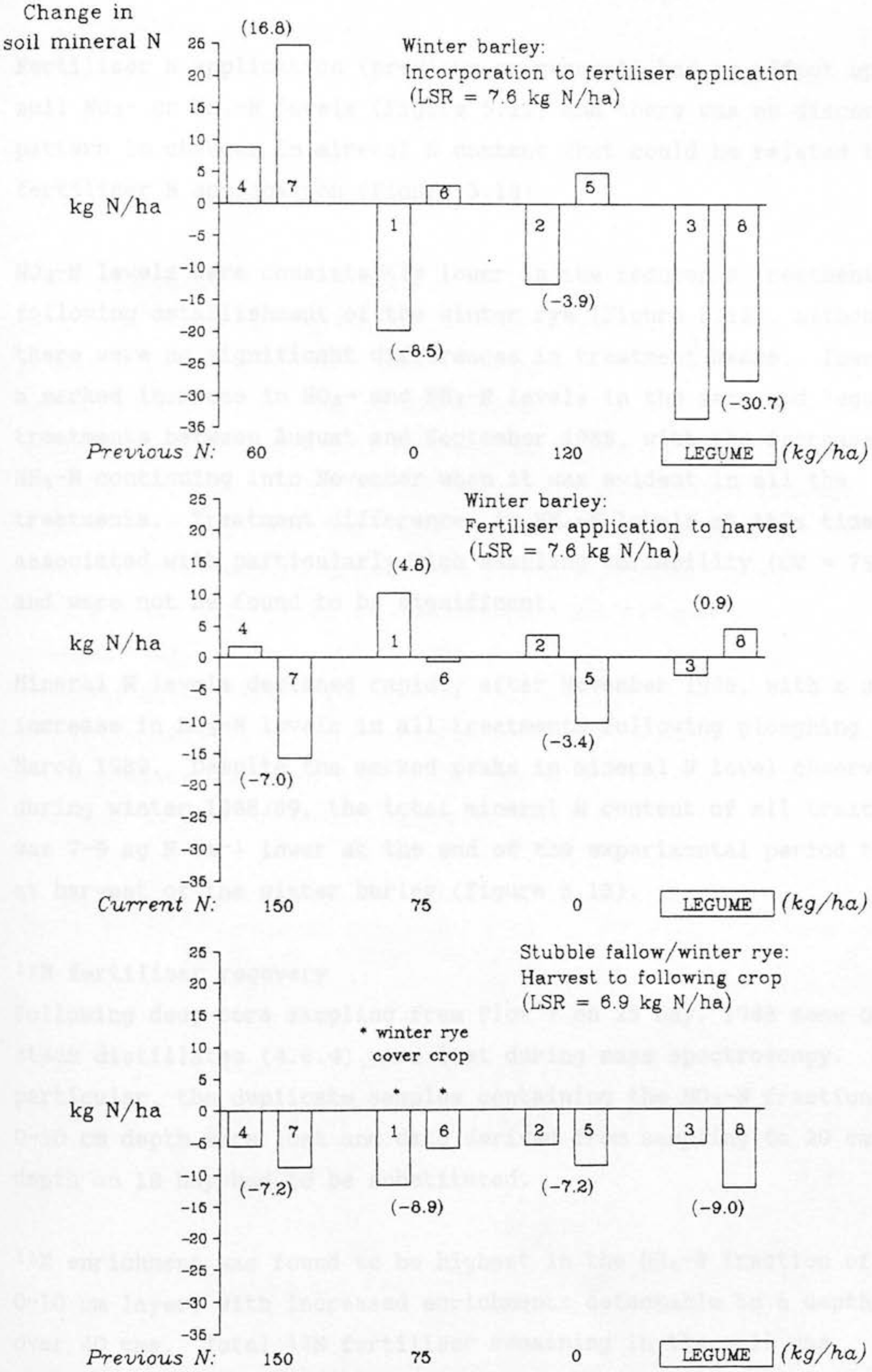
Figure 5.12: Mean soil NO_3-N and NH_4-N levels (mg kg^{-1} soil) in 0-20 cm layer of replicate plots from 4 September, 1987 and 20 April 1989



Further similarities were evident in the zero and legume N plots on their additional sampling dates (4.6.4). The only exception was in September and October 1987 when NO_3-N levels were significantly higher in the legume N treatment (LSR = 9.6, Figure 5.12). Additional data suggested that the increase in NO_3-N in the legume N treatment had begun before incorporation (*i.e.* mean NO_3-N levels at incorporation were 13.0 mg kg^{-1} soil compared to 4.4 mg kg^{-1} soil

Figure 5.13: Change in soil mineral N content (kg ha^{-1}) of plough layer during the three main agronomic periods

Plot numbers are in histograms and treatment means in parentheses



measured at chopping). There was no evidence of a corresponding increase in NH_4-N. By the time of fertiliser N application in March 1988, the mineral N content of the legume N treatment had fallen by over 30 kg N ha^{-1} (Figure 5.13).

Fertiliser N application (previous or current) had no effect upon soil NO_3- or NH_4-N levels (Figure 5.12) and there was no discernible pattern to changes in mineral N content that could be related to fertiliser N application (Figure 5.13).

NO_3-N levels were consistently lower in the reduced N treatment following establishment of the winter rye (Figure 5.12), although there were no significant differences in treatment means. There was a marked increase in NO_3- and NH_4-N levels in the zero and legume N treatments between August and September 1988, with the increase in NH_4-N continuing into November when it was evident in all the treatments. Treatment differences in NH_4-N levels at this time were associated with particularly high sampling variability (CV = 79-83%) and were not be found to be significant.

Mineral N levels declined rapidly after November 1988, with a slight increase in NO_3-N levels in all treatments following ploughing in March 1989. Despite the marked peaks in mineral N level observed during winter 1988/89, the total mineral N content of all treatments was 7-9 kg N ha^{-1} lower at the end of the experimental period than at harvest of the winter barley (Figure 5.13).

^{15}N fertiliser recovery

Following deep core sampling from Plot 7 on 25 May, 1988 some of the steam distillates (4.6.4) were lost during mass spectroscopy. In particular, the duplicate samples containing the NO_3-N fraction from 0-10 cm depth were lost and data derived from sampling to 20 cm depth on 18 May had to be substituted.

^{15}N enrichment was found to be highest in the NH_4-N fraction of the 0-10 cm layer, with increased enrichments detectable to a depth of over 40 cms. Total ^{15}N fertiliser remaining in the soil was estimated to be 4.2 kg N ha^{-1} (s.e. = 0.4 kg N ha^{-1}). When combined with the ^{15}N recovery in leachate (5.3.2) and crop uptake at booting (5.3.4) it was evident that approximately 38 kg of the 75 kg N ha^{-1} applied in the first dressing remained unaccounted for (Table 5.15).

5.3.6 DENITRIFICATION LOSSES

Although there was considerable variation in the N_2O concentrations measured in the sealed chambers of replicate plots, some basic patterns in N_2O emission from the different treatments can be identified (Figures 5.14a and b).

There appeared to be two periods of denitrification activity - autumn 1987 and spring/summer 1988 (no data was collected between November 1988 and April 1989). Increased N_2O concentrations in the autumn were most evident in the legume N treatment (Figure 5.14a), whilst in spring they were very marked in the recommended and reduced N treatments following fertiliser application (Figure 5.14b). Maximum mean N_2O concentrations occurred on 20 April, 1988 (3 weeks after application of the first fertiliser dressing) and were 288 and 172 ppm in the recommended and reduced N treatments respectively (Figure 5.14a). These coincided with the first heavy rainfall after fertiliser application and the peaks in drainflow NO_3-N concentration recorded between 12 - 19 April (5.3.2). Maximum N_2O concentration measured in the legume N treatment in autumn 1987 was 37 ppm (in Plot 3 only).

Table 5.15: ^{15}N balance sheet (kg N ha^{-1}) for Plot 7 between 29 March and 25 May, 1988

				% Recovery:
Leached	:	5.7		7.6
Crop uptake	:	27.0	(6.8)	36.0
Remaining in soil	:	4.2	(0.4)	5.6
		36.9		
Unaccounted for	:		*38.1*	50.8
Total	:	75.0		100.0

() Figures in parentheses are standard errors of means

Figure 5.14a: N_2O concentrations (ppm) and cumulative N_2O flux (ppm) in sealed chambers (with partial acetylene inhibition) in the recommended and reduced N treatments

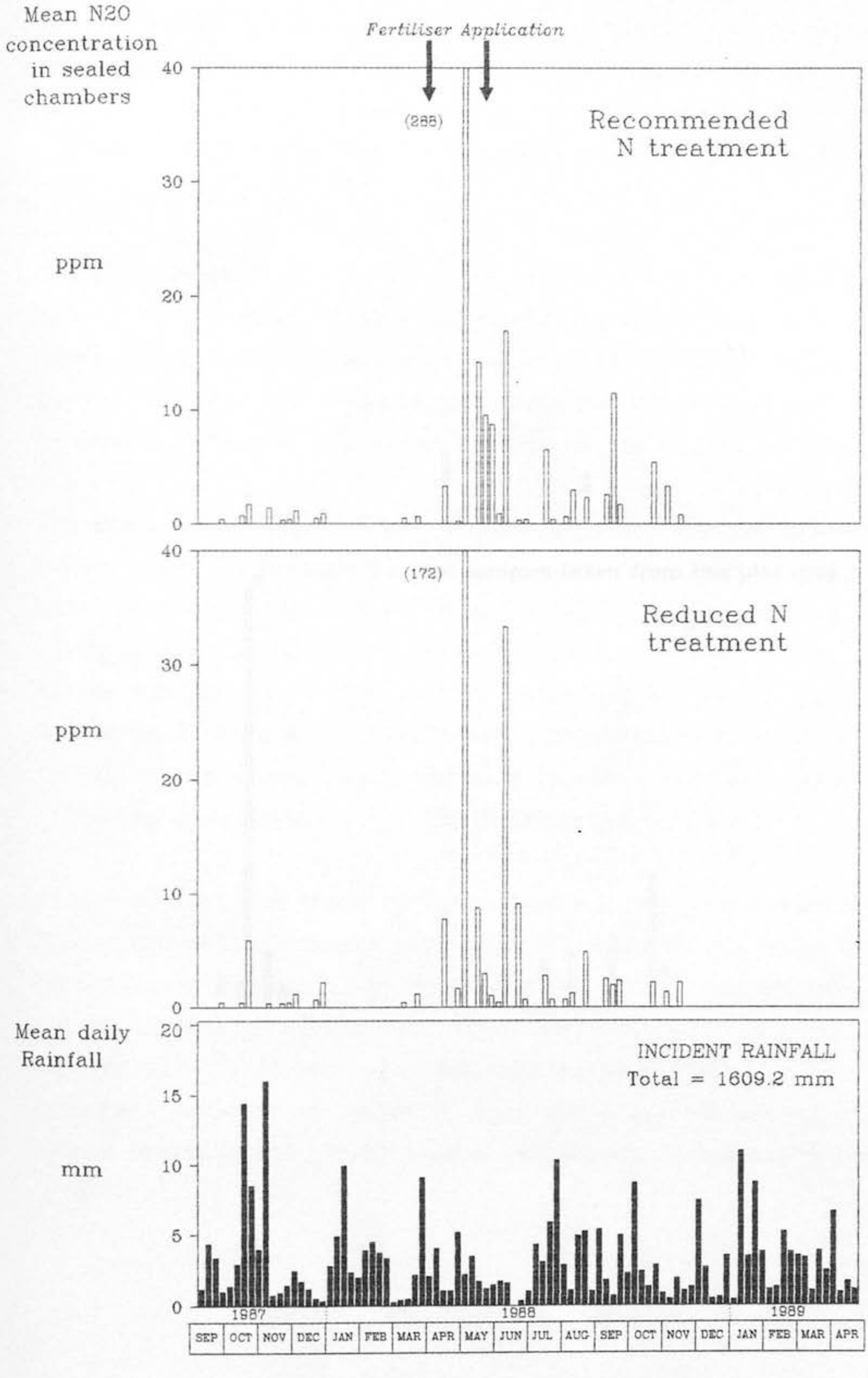
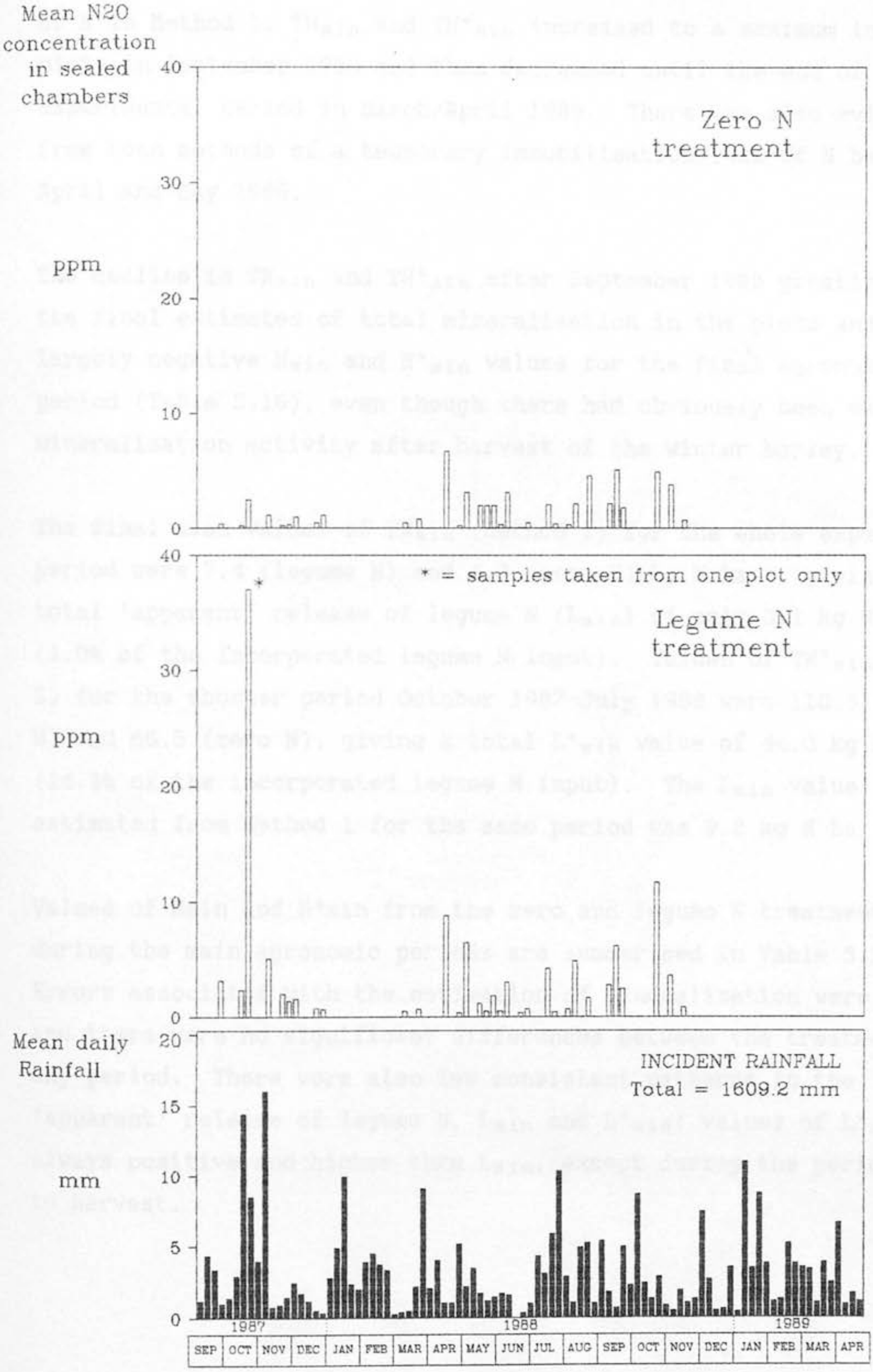


Figure 5.14b: N_2O concentrations (ppm) and cumulative N_2O flux (ppm) in sealed chambers (with partial acetylene inhibition) in the zero and legume N treatments



5.3.7 N RELEASE FROM THE INCORPORATED GREEN MANURE

The pattern of total N release (TN_{min} and TN^*_{min}) estimated by Methods 1 and 2 (4.6.6) was similar in the zero and legume N treatments (Figure 5.15). Excepting an initial immobilisation/loss of N in Method 1, TN_{min} and TN^*_{min} increased to a maximum in all plots in September 1988 and then decreased until the end of the experimental period in March/April 1989. There was also evidence from both methods of a temporary immobilisation/loss of N between April and May 1988.

The decline in TN_{min} and TN^*_{min} after September 1988 greatly reduced the final estimates of total mineralisation in the plots and led to largely negative N_{min} and N^*_{min} values for the final agronomic period (Table 5.16), even though there had obviously been extensive mineralisation activity after harvest of the winter barley.

The final mean values of TN_{min} (Method 1) for the whole experimental period were 7.4 (legume N) and 4.3 (zero N) kg N ha^{-1}, giving a total 'apparent' release of legume N (L_{min}) of only 3.1 kg N ha^{-1} (1.0% of the incorporated legume N input). Values of TN^*_{min} (Method 2) for the shorter period October 1987-July 1988 were 110.5 (legume N) and 66.5 (zero N), giving a total L^*_{min} value of 44.0 kg N ha^{-1} (14.2% of the incorporated legume N input). The L_{min} value estimated from Method 1 for the same period was 9.8 kg N ha^{-1}.

Values of Nmin and N*min from the zero and legume N treatments during the main agronomic periods are summarised in Table 5.16. Errors associated with the estimation of mineralisation were high and there were no significant differences between the treatments in any period. There were also few consistent patterns in the 'apparent' release of legume N, L_{min} and L^*_{min}; values of L^*_{min} were always positive and higher than L_{min}, except during the period prior to harvest.

Figure 5.15: Estimation of total N mineralisation (kg N ha^{-1}) for zero and legume N plots by Method 1 - N recovery and Method 2 - soil incubation

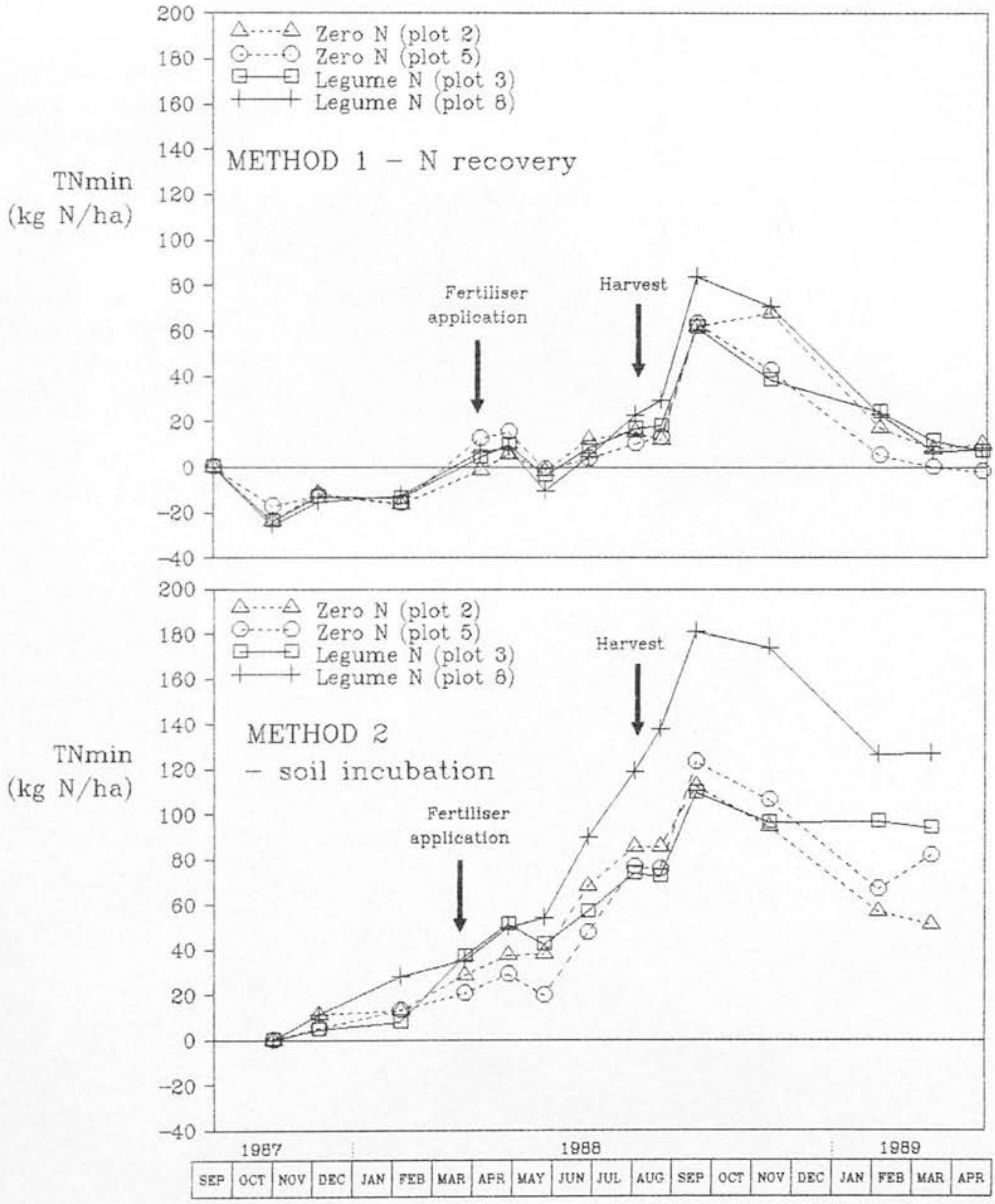


Table 5.16 : Summary of net mineralisation and 'apparent' release of legume N for the main agronomic periods, calculated by Methods 1 and 2

	Method 1 - N recovery			Method 2 - soil incubation		
	Zero	Legume		Zero	Legume	
Period	N_{min}	N_{min}	L_{min}	N^{*}_{min}	N^{*}_{min}	L^{*}_{min}
Green manure incorporation to fertiliser applic.	5.6	5.3	-0.3	24.5[Φ]	36.0[Φ]	11.5
Fertiliser applic. to harvest	7.4	14.5	7.1	56.6	60.5	3.9
Harvest to following crop	-8.8	-12.5	-3.7	-14.6[Φ]	14.0[Φ]	28.6
Total	4.3	7.4	3.1	66.5	110.5	44.0

Φ Incubations were not conducted for full period

The broad aim of this research project was, as outlined in 1.5, to investigate how the N dynamics of a local arable soil with 'conventional' N inputs are modified by a number of different agronomic practices. The experimental approach to this was solely empirical and centred upon the primary objective of directly and accurately measuring NO_3-N leaching losses.

In principle the use of hydrologically isolated field plots was the best approach to quantifying leaching losses under local conditions; an assertion supported at the start of the project by the results of the Boghall Pilot Plot (Chapter 3). In practice the complexity of installing a number of isolated plots, plus the inherently variable nature of glacially-derived soils, contributed to variable drainflow recoveries and hampered the accurate estimation of NO_3-N loadings.

Other workers have experienced similar problems with hydrologically isolated plots and large lysimeters. Hendrick (1921) described at length the problems of establishing large monolith lysimeters on a glacial-till-derived soil in Aberdeenshire and the subsequent problems of obtaining satisfactory drainflow recoveries. More recently, Bergström (1987) found that although the low hydraulic conductivity of a clay soil limited the percolation rate below tile-drained plots, considerable quantities of water were still lost from the plots by deep percolation below the drainage system when groundwater levels dropped. It was also clear that during periods of high precipitation groundwater moved laterally into some plots and contributed to drainflow. Similar problems with groundwater movement were experienced by Harris *et al.* (1984) at Brimstone Farm, with deep water movement apparently occurring beneath the plot isolation barriers. Surface water movement across the permeable fill isolation ditches during major rainfall events was also reported.

Although the drainflow recoveries from Plots 2 and 7 were very encouraging (5.3.1) and indicated the potential for plot isolation on this soil type, the need for correction of NO_3-N leaching losses in the other plots introduced a significant source of error into the results. Whilst care must be taken in interpreting the absolute leaching loss values, the use of a correction procedure should also be seen in perspective:

1. It is likely that denitrification was a more important loss process than leaching on this soil type, but it still remained largely unaccounted for because of the difficulties in measuring it;

2. N dynamics were often dominated by large mineral N fluxes (kg N ha^{-1}) which were comparable to, and frequently exceeded, leaching losses (Figures 5.9 and 5.13). These fluxes were also calculated using an unvalidated correction factor (4.6.4) and were frequently associated with very high sampling variabilities (CV = 21-83%);

3. The results obtained were probably as reliable as any that would have been obtained by other methods (*e.g.* lysimetry, field drain sampling, suction cups or soil sampling), bearing in mind the points above and the practical difficulties of working with this soil type.

The discussion begins by characterising and comparing the dynamics of the Glencorse arable soil receiving recommended and reduced fertiliser N inputs. The processes involved with fertiliser N are well defined and the discussion proceeds by considering many of the processes individually (*e.g.* crop N uptake).

The processes associated with legume N inputs are less well defined and much of the discussion concerns the problems of interpreting the available data in order to identify what processes were occurring after incorporation of the leguminous green manure and how they influenced the availability of legume-derived N to the crop.

6.1 N DYNAMICS OF AN ARABLE SOIL WITH RECOMMENDED AND REDUCED FERTILISER N INPUTS

6.1.1 RAINFALL N INPUTS

The total N input from rainfall was equivalent to an annual deposition rate of approximately 12-13 kg N ha^{-1}. This was similar to estimated annual inputs of 13-25 kg N ha^{-1} (Roberts 1987) and 10-25 kg N ha^{-1} (Central Water Planning Unit 1977) for eastern Britain, and 7.3-15.8 kg N ha^{-1} for the Scottish uplands (Edwards *et al.* 1985).

The mean NH_4-N concentration in rainfall at Glencorse was considerably higher than the typical concentration of 0.36 mg l^{-1} cited for Scotland by Edwards *et al.* (1985). This was probably due to increased atmospheric pollution from local coal burning or industrial activity (Bouwman 1990b). Levels of NO_3-N were very similar to the typical Scottish concentration of 0.46 mg l^{-1}.

6.1.2 CROP YIELDS AND N UPTAKE

Grain yields of the spring and winter barley receiving recommended fertiliser N rates were likely to be typical of much of the poorer quality arable land in Scotland, but were considerably lower than achieved by the same varieties in SAC cereal trials. Even when receiving no fungicide treatment (as at the Glencorse experimental site), the spring barley varieties have been found to yield an average 5.6 - 6.0 t ha^{-1} in trials and the winter barley variety 6.4 t ha^{-1} (Richards 1989). Smith (1988) reported winter barley yields in the Borders of over 10 t ha^{-1}.

As illustrated by Table 6.1, it is difficult to see a reduction in fertiliser N rates being economically viable at Glencorse. Even under the relatively marginal conditions, the two barley crops still showed profitable yield responses to applied N. Only the spring barley at the recommended N rate appeared to be approaching the top of its N response curve and therefore capable of providing a net financial gain through fertiliser N reduction.

The main cause of low crop yields was likely to have been the poor physical condition of the soil, notably the high bulk density and poor drainage of the subsoil (Pidgeon 1980). This may have been exacerbated by plot establishment. The Winton soil series is very susceptible to mechanical compaction (Pidgeon 1980) and it is possible that attempts to minimise traffic over the plots during hydrological isolation were not successful. Vinten *et al.* (1991) reported the presence of a serious pan in the plots during the summer of 1989. This was attributed to cultivations in the preceding spring, but may have been present for considerably longer.

There are a number of ways that adverse soil conditions could have affected yields. Compaction and poor drainage can both be expected to limit root development through increased mechanical impedance (Holmes 1976), oxygen stress (Ellis *et al.* 1984), and the presence of reduction products such as ethylene or hydrogen sulphide (Russell 1973). Holmes (1976) suggested that limited root development in the Winton series results in slower N uptake, which in turn reduces crop growth. This is only true if available soil N is predominantly in the form of NH_4, since this moves relatively slowly to roots (Bock 1984) whereas even restricted root systems can absorb the highly mobile NO_3 ion very effectively (Burns 1980). Reductions in nutrient uptake were therefore likely to have arisen from factors which lowered soil mineral N levels, as well as those that reduced the crop's ability for N recovery. For example, conditions which lead to the occurrence of low oxygen concentrations in the root zone limit root growth, inhibit N absorption and encourage denitrification (Ellis *et al.* 1984).

Table 6.1: Economic response (£ ha^{-1})* of spring and winter barley to reduced fertiliser N application

		Spring barley	Winter barley
Value of fertiliser N saved @ 36p kg^{-1}	:	21.60	27.00
Value of grain yield lost @ £95 t^{-1}	:	13.30	182.40
Net gain/loss (£ ha^{-1})		8.30	-155.40

* based upon 1990/91 prices from SAC (1990)

Vinten *et al.* (1991) further noted that although the easily available water capacity (5-200 kPa) of the soil at Glencorse was large (14.6% for topsoil and 13.5% for subsoil), the restriction on rooting depth meant that the soil was prone to droughtiness in the summer months. This could have contributed to yield reduction in the winter barley during the drier than normal second quarter of 1988 (Table 5.1).

Recoveries of ^{15}N-labelled fertiliser in the spring and winter barley at harvest were within the range of 25-32% for all fertiliser N rates (Tables 5.3 and 5.12). These were low compared to the range of recoveries generally reported for barley (*e.g.* 46-54% by Dowdell *et al.* 1984 in southern England and 43-67% by Nielsen *et al.* 1988 in Denmark), but were similar to values reported for the Winton soil series at other sites in south-east Scotland (Smith *et al.* 1984, 1988). Fertiliser N recovery did not appear to be affected by crop type or N rate, although it has been suggested that the root systems of spring-sown crops (which develop later) are likely to be less effective at absorbing spring-applied N than those of autumn-sown crops (Powlson *et al.* 1986).

^{15}N recoveries in spring barley were not reported in detail by Smith *et al.* (1984), but over 3 seasons appeared to be in the range of approximately 16-46%. In a subsequent paper, the recovery of spring-applied ^{15}N fertiliser in winter barley at a number of sites was reported to be 13-85% over 5 seasons (Smith *et al.* 1988). Recoveries were generally lowest on the Winton series (13-30%), although they also fell drastically on other soil types in the wetter seasons.

Some other characteristics of the crop N data at Glencorse were also noted by Smith *et al.* (1984, 1988) and may have arisen from the poor physical condition of the Winton soil:

1. A 'priming effect' consistently occurred in the spring and winter barley *i.e.* the application of fertiliser N apparently increased the uptake of soil-derived N (Tables 5.3 and 5.12, Figure 5.10). This may also have been reflected in the greater reduction in soil mineral N levels after the recommended fertiliser N application to the winter barley (Figure 5.13).

Smith *et al.* (1984) attributed 'priming effects' in the Winton soil series to increased root exploration, noting that they were not found on a sandy soil which could be expected to display more efficient, unrestricted root exploration. There is, however, considerable controversy over the cause and interpretation of 'priming effects' (*e.g.* Jansson and Persson 1982). In a review of the subject, Jenkinson *et al.* (1985), challenged the idea that fertiliser N can encourage root proliferation, citing a number of studies which indicated that the effect of applied N on root growth was quite small compared with its effect on shoot growth. The authors suggested that 'priming effects' are more likely to be *apparent*, caused predominantly by pool substitution. In other words, a soil fertilised with a labelled N compound accumulates proportionally more unlabelled N than an unfertilised soil because the applied ^{15}N stands proxy for unlabelled N that would otherwise be removed from the mineral N pool by microbial immobilisation or denitrification.

If the 'priming effect' observed at Glencorse was *real* and due to increased root exploration, the application of fertiliser N greatly improved the utilisation of soil-derived N by the spring and winter barley. In the winter barley the efficiency of this soil N use was decreased by reducing N application;

2. In both the recommended and reduced N treatments there was little change in fertiliser N recovery following application of the second split dressing to the winter barley. This contrasts with the marked increase in fertiliser N recovery observed on some soil types and was again attributed by Smith *et al.* (1988) to the poor initial physical condition of the Winton series and its slow improvement during the spring. Other authors (*e.g.* Mary *et al.* 1988) have noted reductions in fertiliser recovery by harvest time and have related it to gaseous N losses from the plant tops or the re-translocation of N to the roots.

Even in the absence of any 'priming effects' (*i.e.* in the zero N treatment) the uptake of soil-derived N was highest in the spring barley crop. This may have been related to greater N mineralisation during the wetter than average 1987 season and/or better utilisation of soil-derived N by the spring barley.

Mean summer temperatures in Scotland are lower than those in the south of Britain and a delay usually occurs in the onset of organic matter mineralisation in spring. N demand by spring barley, which is harvested up to 2 months later than winter barley, is likely to be better synchronised with this natural N supply than winter barley. Smith *et al.* (1984) inferred that under local conditions a large proportion of N uptake by spring barley occurs late in the growing season and is derived mainly from soil N, but there was no direct evidence of this at Glencorse *i.e.* the $\%N_{dfs}$ of the spring barley at harvest was very similar to that at anthesis.

There was evidence of late season N mineralisation following harvest of the winter barley in July 1988. Figure 5.12 shows an increase in soil NO_3- and NH_4-N levels in the zero and legume N treatments between July and September and an increase in NH_4-N levels in all treatments between July and November. There was no evidence of a similar increase in soil mineral N levels after harvest of the spring barley in September 1987.

6.1.3 N LOSSES

Immobilisation by soil micro-organisms, leaching and denitrification are generally cited as the most important processes decreasing fertiliser N availability to plants (Bock 1984). However, they are not the only causes of an apparently low crop fertiliser N recovery.

One limitation of this project was that no account was taken of root N due to the difficulties of effective root recovery and estimation of total root biomass production and N uptake during the growing season (Hansson and Steen 1984). Sampling and separation of roots from the green manure crop (4.6.2) had, for example, proved to be arduous, even though the roots involved were relatively large. Many researchers have therefore simply applied a correction factor to the aboveground crop N data in order to derive a figure for total N uptake.

Rosswall and Paustian (1984) reported on the use of soil cores and estimates of root mortality to calculate a root:shoot N ratio of 0.27 for barley receiving 120 kg N ha^{-1}. The applicability of such a ratio to this work is uncertain since root biomass production and its relationship to aboveground biomass varies with time and environmental conditions, including fertiliser N application (Welbank and Williams 1968, Hansson and Steen 1984).

However, assuming that 0.27 was a valid root:shoot N ratio for all treatments at Glencorse, N uptake in the winter barley roots would have been 20.5 and 10.4 kg N ha^{-1} in the recommended and reduced N treatments respectively. Further assuming that $\%N_{dff}$ of the roots was equal to that of the aboveground crop, approximately 12.8 and 5.4 kg ha^{-1} of fertiliser N was recovered in the roots (7-8% of application).

Immobilisation

According to Jenkinson and Ladd (1981) the soil microbial biomass is not only the mediator of both immobilisation and mineralisation, but also the repository of considerable quantities of N, some of which may be derived from fertiliser.

Shen *et al.* (1989) found the total biomass N content of plots in the Broadbalk Continuous Winter Wheat Experiment at Rothamsted to be in the range of 138 - 210 kg N ha^{-1}. There was no consistent difference in the total biomass N content of plots receiving 48 - 192 kg fertiliser N ha^{-1} $annum^{-1}$ and no systematic variation in the immobilisation of fertiliser N with increasing application rate. At harvest, the microbial biomass contained 3-8% of ^{15}N-labelled fertiliser applied 4 months earlier (equivalent to 19-27% of the total ^{15}N remaining in the soil). Neeteson *et al.* (1986) suggested that the N uptake by microbial biomass immediately following fertiliser application can be significantly higher than this, leading to the apparent 'disappearance' of up to 85% of applied N. Most of this was re-mineralised during the subsequent 5 weeks.

There is little information available on the significance of soil microbial biomass in local soils or the role it might play in reducing fertiliser N recovery. Smith *et al.* (1985) noted some crop uptake of

^{15}N from labelled fertiliser applications in previous years and attributed it to the immobilisation and subsequent re-mineralisation of labelled-fertiliser N. Rees (1989) used a fumigation technique to measure microbial biomass N in field and laboratory incubated soil cores taken from the Winton soil series. An increase in the N content of soil biomass was noted following NO_3- and NH_4-N application, and it was concluded that soil biomass can be an important sink for fertiliser N under local conditions.

The potential size of this sink remains unknown. Data reported by Powlson and Jenkinson (1981) on the biomass carbon content of soils on the Bush Estate near Edinburgh, in North Yorkshire, Cambridgeshire and at Rothamsted suggested that microbial biomass levels in Scotland may be lower than elsewhere in the UK, even though soil organic matter levels tend to be higher. Work by van der Linden *et al.* (1989) further indicated that soils with naturally high bulk densities, or those suffering from compaction, display lower microbial activity and a lower turnover rate of organic materials through the soil biomass.

Leaching

NO_3-N concentrations at Glencorse were generally lower than those from experiments in southern Britain and this was probably related in part to the 'dilution' effect of the higher annual rainfall and soil drainage discharge. For example, long-term annual rainfall on the Bush Estate was 866 mm, compared to 680 mm at Brimstone Farm (Cannell *et al.* 1984); whilst normalised winter drainflow (Plots 2 and 7) in 1987/88 and 1988/89 was approximately 420 mm compared to 73 - 227 mm over 8 years at Brimstone Farm (Goss *et al.* 1988).

Annual leaching losses from the recommended and reduced N treatments were equivalent to 21.9 - 22.5 and 14.8 - 21.5 kg N ha^{-1} respectively. Estimates of annual loss varied depending upon whether the 'year' was taken from incorporation of the green manure to harvest of the winter barley, or fertiliser application to establishment of the following crop. These losses were slightly higher than measured in north-east Scotland and generally less than measured in southern Britain. Subsequent work at the site showed higher losses after a dry summer (Vinten *et al.* 1991).

In Aberdeenshire, Edwards *et al.* (1990) estimated annual losses of 15 kg N ha^{-1} $annum^{-1}$ in a mixed cropping catchment study, while over 50 years earlier (Hendrick and Welsh 1938) reported losses of 1.0 - 11.4 kg N ha^{-1} from barley and oat crops in mixed rotation on the Craibstone lysimeters. In southern Britain, mean annual losses from winter wheat and oats at Brimstone Farm were 34 kg N ha^{-1}, with a range of 3.2 - 75.3 kg N ha^{-1} between 1980 and 1984 (Dowdell *et al.* 1987). These compared closely with losses of 41 kg N ha^{-1} from the lysimeter study of Webster *et al.* (1986), but were lower than the losses of 65 - 83 kg N ha^{-1} reported by Dowdell *et al.* (1984) from spring barley grown in lysimeters. Mean annual losses of 34.0 and 38.6 kg N ha^{-1} were obtained from mixed cropping catchment studies in eastern and south-western England respectively (Roberts 1987, Burt and Arkell 1987).

Gustafson (1987) found a similar north-south trend in the regional distribution of leaching losses from arable soils in Sweden. Monitoring of field drains at four sites over 13 years revealed mean NO_3-N losses from cereal crops to be highest (32.1 kg N ha^{-1} $annum^{-1}$) in the far south of the country. Although there was some interaction with fertiliser N rates and the residual effect of preceding crops, the author largely attributed the results to the milder climate, and therefore greater mineralisation potential, of the southern most site.

A number of other points also emerged from the Glencorse data:

1. There was no evidence from either winter period of any relationship between NO_3-N leaching losses and fertiliser application to the preceding crop. This suggests that soil-derived N rather than unused fertiliser-N was the most important source of winter NO_3-N leaching.

   Similar conclusions have been reached by many other workers. Dilz (1988), reviewing recent work, showed that N fertilisers applied at rates up to the economic optimum left little more inorganic N in the soil than was present in unfertilised controls (the mineral N data from Glencorse also showed this). ^{15}N work by Macdonald *et* *al.* (1989) at Rothamsted and Recous *et al.* (1989) in France consistently found that less than 2% of fertiliser N applied to a winter cereal in spring remained in the soil as inorganic N at harvest;

2. The loss of 7.6 - 14.1 kg N ha^{-1} during both winters was similar to losses from the Boghall Pilot Plot (3.2.2), but again lower than reported winter losses from further south. Approximately 18 - 21 kg N ha^{-1} were lost from plots under winter fallow at Cockle Park in Northumberland (Armstrong *et al.* 1983), while winter losses at Brimstone Farm were 43.6 and 59.7 kg N ha^{-1} in the first two years (1978-80) of the experiment (Harris *et al.* 1984) and 2.7 - 46.3 kg N ha^{-1} for the following six years (Goss *et al.* 1988);

3. There was little difference in leaching losses from winter barley and stubble fallow in the successive winters at Glencorse ('expected' drainflows were very similar). This was contrary to the usual observation that leaching losses are reduced by autumn-sown crops (Powlson 1988) and was probably related to increased losses following ploughing and establishment of the winter barley.

   Results from Brimstone Farm showed greater leaching losses from ploughed soil than direct-drilled soil (Goss *et al.* 1988). This was attributed to the soil disturbance during cultivation enhancing the mineralisation of soil organic matter and increasing quantities of available NO_3-N at a time when the crop was unable to utilise it (*e.g.* Dowdell *et al.* 1983).

   Whilst the 1987 autumn peaks in drainflow NO_3-N concentration at Glencorse (up to 8.9 mg l^{-1}) indicated the occurrence of enhanced late season mineralisation, they were small compared to Brimstone Farm where peaks in the range of 50 - 95 mg l^{-1} occurred and NO_3-N levels remained substantially above the EEC limit of 11.3 mg l^{-1} for the first 100 mm of winter drainflow (Harris *et al.* 1984, Dowdell *et al.* 1987). Roberts (1987) also found NO_3-N levels in catchment outflow regularly exceeded 11.3 mg l^{-1} for 1 - 2 months every autumn, often reaching 30 - 40 mg l^{-1};

4. Although NO_3- and NH_4-N levels in the soil profile increased during autumn 1988 (Figure 5.12), there was no evidence of increased NO_3-N concentrations in the drainage water. Reasons for this are not clear, but denitrification may have been a more important loss process in the winter of 1988/89 due to higher soil temperatures (Figure 5.1) and more conducive soil conditions in the unploughed stubble fallow. Colbourn (1988) reported that denitrification rates were higher in a direct-drilled soil than a ploughed one and suggested this might be related to the undisturbed soil structure;

5. Leaching losses from UK arable agriculture are generally considered to be more important during the winter than in the spring (Addiscott 1988). At Glencorse the total amounts of NO_3-N lost in winter and spring were similar, since although spring drainflow was lower NO_3-N concentrations were higher;

6. Significant increases in drainflow NO_3-N concentration during the spring of 1987 and 1988 suggested the susceptibility of applied fertiliser N to direct leaching losses. The spring NO_3-N peaks (up to 58 mg l^{-1}) were much higher than those occurring in the autumn and were of a similar size to those reported in southern Britain following fertiliser application; for example, up to 60 mg NO_3-N l^{-1} from the Brimstone plots (Dowdell *et al.* 1987) and up to 70 mg l^{-1} in tile drains at Saxmundham in Norfolk (Williams 1970).

   Goss *et al.* (1988) stressed the variable nature of spring leaching losses, pointing out that they were closely related to rainfall in the period following fertiliser application. Spring leaching losses were high at Glencorse because heavy rainfall occurred 2-3 weeks after the first split fertiliser dressing, at a time when the soil was still well above field capacity. ^{15}N data confirmed that 86% of the NO_3-N leached from Plot 7 during this heavy rainfall was derived from applied fertiliser, much of which was probably lost via rapid 'bypass' flow from the soil surface to the base of the plough layer (Vinten and Redman 1990).

   In total, 7.6% of the ^{15}N labelled-fertiliser applied to Plot 7 at the first split dressing was leached by the time of the second dressing, with little evidence of any further ^{15}N leaching after this. This is very different to the pattern of loss reported from studies where spring applications of ^{15}N labelled-fertiliser have been made to soils below field capacity.

Dowdell *et al.* (1984) and Bergström (1987) applied single dressings of 80 - 120 kg labelled fertiliser N ha^{-1} to spring barley in lysimeters. They reported the recovery of small amounts of ^{15}N immediately that drainage re-commenced in the autumn, but total recovery in drainage water remained very low and occurred over several years indicating that it was via microbial immobilisation and re-mineralisation. Bergström (1987) only recovered 0.4 - 1.2% of applied ^{15}N over 3 years, while Dowdell *et al.* (1984) recovered 6.3 - 6.6% over 4 years.

7. Reducing fertiliser N application to the spring barley decreased the peak NO_3-N concentrations recorded in spot samples during 1987, suggesting that leaching losses might be related to fertiliser N application. In the winter barley there was no apparent effect of fertiliser N rate on spring leaching losses and no significant difference between the fertiliser and zero N treatments.

   Any treatment effects may, however, have been masked by soil variability between plots. Although ^{15}N data proved the occurrence of fertiliser N leaching, total losses from Plot 7 during the spring period were slightly less than from the zero N treatment in Plot 2. Reasons for this are not clear. Both plots appeared to have no hydrological problems, but mean NO_3-N concentrations were consistently higher from Plot 2 throughout the experimental period and this may have indicated a higher mineralisation potential;

8. Although the winter growth and N uptake of the winter rye was comparable to the winter barley in the previous year, it was limited compared to potential growth in southern Britain where autumn/winter N uptakes of 60 kg ha^{-1} have been reported (Christian 1990). The winter rye did, however, slightly decrease soil mineral N levels and winter leaching losses, thus confirming other work on the efficacy of winter cover crops (*e.g.* Bertilson 1988, Nielsen

and Jensen 1985). Considerably more work is needed on cover crops under local circumstances, including their efficacy compared to weed cover. For example, weed growth at Glencorse was quite extensive and in the limited number of plots sampled showed high dry matter accumulation.

Leaching losses from the Winton soil under 'conventional' arable production did not therefore appear to present a serious environmental/public health risk, particularly when it is borne in mind that 3 - 4 kg ha^{-1} of the annual leaching losses were derived directly from NO_3-N in rainwater.

Losses of soil-derived mineral N in the autumn were less important than in southern Britain, although cultivation did accelerate the mineralisation of soil organic matter and increase NO_3-N losses. NO_3-N concentrations only exceeded the EEC limit of 11.3 mg l^{-1} for 1 - 2 weeks in the spring and this was due to the greater susceptibility of fertiliser N to direct leaching loss when applied to soils above field capacity.

There was no evidence to indicate that reducing fertiliser N application would decrease spring leaching losses and, since there was little inorganic fertiliser N remaining in the soil after harvest, no evidence to suggest it would decrease winter losses. Indeed, it is possible, given the strong 'priming effect' that fertiliser N apparently had upon crop uptake of soil N, that a reduction in fertiliser N would increase winter leaching losses by increasing the levels of soil N remaining at the end of the growing season.

Accepting that spring leaching losses are variable and may be unavoidable on this soil type, the most promising option for reducing leaching losses from the Winton soil under conventional arable production might be to modify the cropping to make better use of soil-derived N and avoid its winter loss as much as possible. For example, based upon the results of this project, growing spring barley undersown with an over-wintering crop such as Italian ryegrass would:

a) make better use of soil N from late season mineralisation;

b) avoid the risk of leaching associated with autumn cultivations;

c) help reduce winter leaching losses by maintaining crop cover.

Denitrification

The Winton soil series is presumably very susceptible to denitrification; it is poorly structured, imperfectly drained and heavily gleyed, suggesting that anaerobic conditions are widespread (Smith 1977) and unlikely to be limiting to the denitrification process.

As already noted (4.6.5) the absolute measurement of denitrification losses by the acetylene inhibition technique is largely defeated on the Winton soil by the very slow and incomplete diffusion of acetylene into the soil profile (Arah *et al.* 1991). The significance of this problem is likely to be exacerbated by the increased reduction of N_2O to N_2 under conditions of high soil water content and poor soil structure (Arah and Smith 1990). Measured N_2O fluxes from the Winton soil have accordingly been found to be at least one order of magnitude *less* than fluxes from lighter soils which would otherwise be expected to display considerably lower denitrification activity.

Despite these limitations increased N_2O concentrations were measured in the sealed chambers at Glencorse and generally agreed with the typical denitrification pattern described for arable clay soils by Colbourn (1988) *i.e.* autumn activity, followed by a mid-winter lull and then resurgence of activity in the spring coinciding with fertiliser application. In particular, the heavy rainfall after fertiliser application in April 1988 greatly increased N_2O concentration in the fertiliser N plots, suggesting a significant denitrification (as well as leaching) loss of fertiliser N at this time.

Having identified that denitrification activity occurred in the fertiliser N treatments, and that it was largely associated with spring N application, estimates of total denitrification loss can be derived from the available ^{15}N data (*e.g.* Dowdell and Webster 1984) by assuming that:

a) the only significant denitrification loss was that of fertiliser N;
b) the whole of this loss occurred within a relatively short period after application;
c) the ratio of ^{15}N applied:^{15}N denitrified was the same as total N applied:total N denitrified.

Total ^{15}N unaccounted for between the first and second fertiliser application to Plot 7 was 38.1 kg N ha^{-1}, 51% of that applied (Table 5.15). If this was all due to denitrification, *maximum* loss from the recommended and reduced N treatments was 76.5 and 38.3 kg N ha^{-1} respectively. Estimation of *minimum* losses must take account of root ^{15}N uptake and possible immobilisation. For example, by assuming:

i) *up to* 12.8 and 5.4 kg N ha^{-1} of fertiliser N was recovered in crop roots in the recommended and reduced N treatments respectively (as estimated above);
ii) *up to* 8% of applied fertiliser N was immobilised (Shen *et al.* 1989) *i.e.* 12.0 and 6.0 kg N ha^{-1} in the recommended and reduced N treatments respectively.

This suggests minimum losses of 51.7 and 26.9 kg N ha^{-1} from the two treatments, or about 35% of applied N. This is slightly higher than the 25-30% of fertiliser N reported to be lost when soils remain saturated for several days soon after application (Goulding and Colbourn 1988).

The absolute validity of these indirect estimates of denitrification is questionable. They are greatly limited by the lack of information on root N uptake and microbial immobilisation, as well as by ignoring the denitrification losses occurring at other times of year. However, the ^{15}N balance approach probably gives a more realistic indication of the true size of total denitrification losses (N_2O and N_2) on the Winton soil, than that currently provided by field- and laboratory-based techniques (Arah *et al.* 1991).

6.2 N DYNAMICS OF AN ARABLE SOIL WITH A LEGUME N INPUT

6.2.1 N_2 FIXATION AND THE LEGUME N INPUT

Total N accumulation of the forage peas during the 16 week growth period was very high, and the 314.2 kg N ha^{-1} derived from N_2 fixation was the largest single N flux measured at the experimental site. Comparison with other work on forage and field peas can only be made on the basis of the above-ground crop, since this is the crop fraction most commonly investigated (particularly in agronomic trials). However, as this work confirms the total N yield of roots represents a significant proportion (≈ 14%) of total legume N (*e.g.* Bergersen and Turner 1983).

The forage peas showed a higher above-ground dry matter (DM) yield than normally reported in Scotland *i.e.* 9.6 t ha^{-1} (Table 5.4), compared to 5.5 - 8.5 t ha^{-1} reported from trials in the west of Scotland (Potts 1980,1982). This may have been related to a number of factors at the Glencorse site, including the wetter than average spring, later sowing date (Lockhart and Richards 1980), higher seed rate (Potts 1980) and application of 20 kg "starter N" ha^{-1} at emergence (Jensen 1986b).

DM yield was, however, relatively low compared to the 12.2 t ha^{-1} reported by Jensen (1987) for field peas in Denmark. Although, total N accumulation was also high in the Danish crops, levels of fixed N were smaller due to lower estimates of $\%N_{dfa}$ *i.e.* 44 - 79% over 4 seasons (Jensen 1986a, 1987), compared to 89% at Glencorse.

As discussed in 2.2, the absolute validity of the isotope dilution technique may still be questionable, but the value of $\%N_{dfa}$ calculated in this work was felt to be reasonably accurate, since:

a) use of the inter-cropped barley as a reference overcame the problem of soil variability at the site (*e.g.* soil mineral CV = 21-83%) and this is likely to have greatly increased accuracy (Reichardt *et al.* 1987). Co-efficients of variation for $\%N_{dfa}$ in the different crop fractions were in the range of 4.6 - 11.9%;

b) accuracy might have been limited by the dilution effect of N transfer from the peas to the inter-cropped barley during the growing season, but there was no evidence of this occurring *i.e.* ^{15}N atom% excesses of the rape and barley in the guard areas were consistently lower than the inter-cropped barley;

c) errors associated with site heterogeneity, mismatch between reference and fixing crop, or the contribution of seed-borne N to crop uptake become smaller as the amount of fixation increases (Danso 1986, Reichardt *et al.* 1987).

Total N_2 fixation is closely related to the levels of mineral N present in the soil during mid- to late-season when potential fixation activity is at a maximum: as mineral N levels increase, N_2 fixation decreases (*e.g.* Jensen 1986b). It is possible therefore that fixation was promoted at Glencorse by the inherently low soil fertility. It was evident, for example, that uptake of soil and "starter" fertiliser N by the peas took place mainly during the first 50-60 days of growth (Figure 5.4), with the maximum daily fixation rate occurring after 70 days (Figure 5.5). The inter-cropped barley may also have stimulated N_2 fixation by increasing competition for the available soil mineral N (*e.g.* Danso *et al.* 1987).

Despite the high levels of N_2 fixation achieved, the forage peas did not represent a cheap source of fixed N. A conservative estimate of the total cost (*i.e.* just fertiliser and seed) of the 314.2 kg N fixed ha^{-1} was £127 (SAC 1990). This was approximately 40p kg^{-1} of fixed N, compared to the quoted price of 36p kg^{-1} for fertiliser N (SAC 1990). When the opportunity cost (*i.e.* lost gross margin) of the cash crop displaced by the green manure is also considered, the legume N input was very expensive.

6.2.2 UTILISATION OF THE LEGUME N INPUT

Utilisation of the legume N input by the winter barley crop was very poor and using it to replace the recommended fertiliser N application decreased barley yield by 60%. As Table 6.2 indicates, the use of legume N was therefore economically unviable even *before* the cost of the green manure was taken into account (6.2.1).

It is generally acknowledged that legume residues are a less efficient N source than fertilisers (*e.g.* Smith *et al.* 1987). However, the significant yield reduction observed at Glencorse contrasts sharply with other research work. Dyke *et al.* (1977) reported relatively small reductions in yield (up to 25%) when replacing fertiliser with legume N and in some cases no yield reduction at all (Figure 1.2). In North America, Rohweder *et al.* (1977) reported that maize following a ploughed-in legume always outyielded conventionally fertilised maize, regardless of how much N was applied (up to 300 kg ha^{-1}). Baldock and Musgrave (1980) and Groffman *et al.* (1987) concluded that legumes had the potential to produce large amounts of N for crop production without environmental risk or decreases in soil fertility.

The 'apparent' recovery of legume N in the winter barley at Glencorse was only 4.2% (although tentative use of ^{15}N data suggested an uptake of 7.3%) and considerably lower than recoveries reported by most other workers. For example, Ladd et al. (1981a and 1983) recovered 11 - 28% of labelled lucerne N in first-year wheat crops, while Müller (1988) found the uptake of labelled clover N by barley was 11 - 20% of the input.

Table 6.2: Economic response (£ ha^{-1})* of winter barley to replacing recommended fertiliser N application with a source of legume N

		Winter barley
Value of fertiliser N saved @ 36p kg^{-1}	:	54.00
Value of grain yield lost @ £95 t^{-1}	:	269.80
Net gain/loss (£ ha^{-1})	:	-215.80
Cost of green manure (£ ha^{-1})	:	127.00
TOTAL COST (£ ha^{-1})	:	-342.00

* based upon 1990/91 prices from SAC (1990)

Identifying reasons for such low utilisation of legume N is difficult since, as already discussed (1.4.2), the N dynamics involved are complex. The fate and plant availability of legume N is largely determined by the interaction of residue decomposition and N mineralisation with other N cycle processes. According to Smith *et al.* (1987), this can often lead to problems of "poor synchronisation" between N release (supply) and crop N uptake (demand) which contributes to reduced utilisation of the legume N.

There are a number of cited examples of this "poor synchronisation." For example, Huntington *et al.* (1985) reported that N release from decomposing vetch residues in North America occurred too late in the season for utilisation by maize at its critical growth stage. The "poor synchronisation" of legume N release at Glencorse appeared to involve two specific elements: an *oversupply* of N in autumn 1987 followed by an *undersupply* in spring 1988.

1. **Autumn 1987**

   Increases in soil mineral N prior to incorporation of the green manure suggested that decomposition of the legume material began whilst it lay on the soil surface after chopping. This is a common observation, although mineralisation rates do tend to be much slower on the surface, particularly during the summer (Smith *et al.* 1987). There may also have been some potential for NH_3 volatilisation from the legume residues whilst on the surface. This has not been extensively investigated, but Kirchmann (1985) reported large amounts of NH_3 released from low C:N ratio green manure material decomposing aerobically in the absence of soil.

   The shallow incorporation of the green manure in early September 1987 (followed by ploughing and seed bed preparation in late September) increased soil NO_3-N levels significantly, suggesting a very rapid initial release of N from the legume material. This is typical of most organic residues (Jenkinson 1981), although it is generally accepted that leguminous residues decompose at a much faster rate due to their low C:N ratio and/or plant composition. Powlson (1980) also noted that cultivations may accelerate the decomposition of fresh organic matter more than older, stable organic matter.

Initial incorporation was over 3 weeks before the crop was drilled and well over 6 weeks before it had fully emerged. This provided an ideal 'window' for autumn leaching and over 20 kg NO_3-N ha^{-1} were lost in drainage water before the end of October (over 50% of the *total* leaching losses from the legume N treatment during the whole experimental period). Significant leaching losses from autumn-incorporated legume residues, including ploughed grass/clover leys, have been reported by numerous workers (*e.g.* Adams and Pattinson 1987, Davies and Barraclough 1988) and all illustrate the difficulty of synchronising autumn legume N release with crop N requirement.

It is not possible to say exactly how much legume N was released in autumn 1989. Soil NO_3-N levels dropped rapidly after their initial peak and estimated values of TN_{min} over the first 50 days after incorporation showed a net loss of N from the system rather than a net gain (Figure 5.15). This suggested significant denitrification losses immediately after incorporation of the green manure. These were confirmed by increased N_2O emissions in the sealed chambers on the legume N plots and in laboratory incubated cores taken at the site by Arah *et al.* (1991). It is possible that the rapid nitrification of legume-derived N during this period may also have contributed to N_2O emissions (Bouwman 1990b).

Denitrification is known to be stimulated by a wide range of organic compounds, including leguminous plant residues (Smid and Beauchamp 1976). Aulakh *et al.* (1983), for example, reported significant N_2O losses after the incorporation of clover as a green manure and Groffman *et al.* (1987) suggested that legume N inputs may be more susceptible to denitrification losses than fertiliser N inputs. Goulding and Webster (1989) discussed a number of reasons why incorporated leguminous material is likely to increase denitrification rate. Firstly, legumes contain more readily-available, water-soluble carbon than other plant residues and therefore stimulate rapid increases in microbial biomass leading to a larger denitrifying population. Secondly, the decomposition and mineralisation of the low C:N ratio legume material increases the availability of NO_3-N for denitrification. Finally, there is the general effect of 'fresh', decomposing organic material which creates localised anaerobic zones and so-called 'hot-spots' of denitrifying activity.

2. **Spring 1988**

   In contrast to the relatively high availability of legume N immediately after incorporation, there was a very low release of legume N for the remainder of the experimental period.

   For example, the estimated N release (L_{min} and L^*_{min}) from the incorporated legume material between the time of fertiliser application and harvest was only 7.1 and 3.9 kg N ha^{-1} respectively (Table 5.16). L_{min} was undoubtedly an underestimate since it did not include N uptake by the crop roots (4.6.6). In the subsequent preparation of total N balances (6.3) an additional 3.5 kg ha^{-1} of legume N was estimated to have been recovered in roots by harvest (Table 6.3). The estimation of L^*_{min} may also have been subject to error since, despite the simplicity of the method, artefacts may have been introduced which altered the rate of mineralisation or other N transformations in the incubated cores. These artefacts include (Raison *et al.* 1987):

   - enhanced mineralisation due to disturbance of the soil structure and/or maintenance of a favourable soil moisture regime (moisture content at sampling is maintained throughout the period of incubation);
   - enhanced denitrification losses caused by NO_3-N accumulation and maintenance of an unfavourable soil moisture content regime;
   - increased immobilisation caused by higher mineral N concentrations and the presence of freshly excised roots (*i.e.* altering the availability of carbon).

Rees (1989) found relatively limited accumulations of mineral N in field-incubated cores in the Winton soil and attributed this to re-immobilisation of mineralised N (an increase in microbial biomass was noted in the same cores).

Although the absolute validity of the estimated values of L_{min} and L^*_{min} at Glencorse was questionable, it still remained obvious from crop growth in spring 1988 that N supply from the incorporated legume was considerably less than from fertiliser application. For example, the 'N fertiliser equivalence' of the legume N treatment, as calculated from the crop's N response curve at harvest, was less than 40 kg N ha^{-1}.

A number of factors may have contributed to the low 'apparent' mineralisation of the legume material during the spring and summer of 1988. Firstly, there were those that reduced the gross rate of decomposition and mineral N release; and secondly, those that limited the availability of this mineral N for crop uptake:

a) Low soil temperatures would have reduced the decomposition rate of the legume material (*e.g.* Ladd *et al.* 1985) and delayed the onset of mineralisation until later in the season. Late season N mineralisation was evident in all the plots following harvest of the winter barley (Figure 5.12), but there was no indication that rates were any higher in the legume N treatment than in the zero N (Figure 5.15);

b) Poor soil physical conditions may also have had an effect on decomposition, particularly through reduced soil aeration (*e.g.* Jenkinson 1981). However, very little is known about the specific effects of soil type on legume N release. In Australia, Ladd *et al.* (1981a) reported significantly lower decomposition rates of medic (*Medicago littoralis*) in heavy clay soils during the first 16 weeks after incorporation, followed by similar rates in all soils investigated. Müller (1988) found that soil type affected the release of clover N only slightly in Southern Finland;

c) The incorporation of the legume material may have continued to enhance denitrification losses during the spring and summer of 1988 (N_2O emissions from the legume N plots were generally higher than in the zero N plots). At Rothamsted, Goulding and Webster (1989) found significant denitrification losses (up to 2 kg N ha^{-1} day^{-1}) during early summer on trial plots that had received large inputs of organic matter from farmyard manure and ploughed leys the previous season. They attributed some of the denitrification activity to long-term increases in total carbon content in the plots, but also noted the importance of 'hot spots' of activity arising from lumps of FYM and ploughed-out ley that still remained in the soil and were creating local anaerobic pockets;

d) Even though legume residues have a low C:N ratio, it can be expected that a large fraction of their N content released upon decomposition will be incorporated directly into the microbial biomass and retained in the soil rather than made immediately available for crop uptake (Smith *et al.* 1987). Ladd and Amato (1986), for example, applied ^{15}N-labelled fertiliser and legume material to soils sown with wheat crops. Crop uptake of ^{15}N from both the fertiliser and legume was directly related to respective N inputs, but the percentage recovery of the legume N in the soil organic fraction was over twice that of the fertiliser N.

In some cases, net immobilisation of soil N after legume incorporation has been reported. Ladd *et al.* (1986) reported net immobilisation of soil N for up to 8 weeks after incorporation of senesced legume material in the field and the results of Frankenberger and Abdelmagid (1985) suggested some immobilisation of soil N during the laboratory incubation of leguminous plant stems, even after 20 weeks. In pot experiments, Azam *et al.* (1985) found the addition of legume material reduced the uptake of ^{15}N-labelled NH_4SO_4 and Smith *et al.* (1989) noted the rapid disappearance of NO_3- and NH_4-N when incubated with dried forage pea material taken from the Glencorse plots.

Identifying the relative significance of these factors for legume N availability in spring and summer 1988 is not possible without further investigation. However, it was clear from the following observations that enhanced denitrification and/or immobilisation did occur, and that this not only limited legume N availability, but also that of soil-derived N:

a) tentative ^{15}N data for the legume N treatment suggested a negative 'priming effect' in the winter barley crop *i.e.* the application of legume N apparently decreased the uptake of soil-derived N;
b) spring leaching losses from the legume N plots were lower than from the zero N plots (Figure 5.9);
c) there was a slight check in crop N uptake from the legume N treatment between March and April (5.3.4);
d) estimated mineralisation values ('N recovery' and soil incubation) showed a net loss of 10-20 kg N ha^{-1} from the zero and legume N treatments between April and May.

It seems reasonable to conclude from the preceding discussion, that the growth and incorporation of a leguminous green manure is of little value for supplying large amounts of available N for subsequent crop uptake. Instead the value of decomposing legume residues is likely to be the long-term maintenance of soil organic N status *i.e.* as concluded by Müller and Sundman (1988) and Ladd *et al.* (1981b).

6.2.3 LEACHING LOSSES

Although utilisation of the legume N input was very low, this was not reflected in greatly increased total NO_3-N leaching losses. Annual leaching losses from the legume N treatment were equivalent to 9.4 - 33.1 kg N ha^{-1} depending upon whether the first autumn period was included (Figure 5.9). These were low compared to other work. Adams and Pattinson (1985) reported leaching losses of approximately 90 kg N ha^{-1} after the incorporation of a white clover ley and 60 kg N ha^{-1} after incorporating arable pea residues, while Bergström (1987) measured losses of up to 42 kg N ha^{-1} in the first 20 weeks after incorporation of a grass ley. Davies and Barraclough (1988) measured losses of 99 kg N ha^{-1} after ploughing of a grass/clover ley on an organic farm in southern England.

The 'apparent' leaching loss of legume-derived N (calculated by difference) during the entire experimental period was only 14.3 kg N ha^{-1}, which was less than 5% of the incorporated legume N input. This was higher than the leaching loss reported by Müller (1987) using ^{15}N-labelled clover buried in lysimeters, but still suggested that legume-derived N was not very susceptible to leaching loss despite the initial very rapid loss in autumn 1987. The most interesting feature of leaching losses from the legume N treatment was that, with the exception of the first autumn, they remained consistently lower than the zero N treatment. As already indicated this was probably due to increased immobilisation/denitrification of soil-derived N in the presence of the legume material.

6.2.4 IMPROVING THE UTILISATION OF THE LEGUME N INPUT

Some improvements in the utilisation of autumn-released N may have been possible by delaying chopping and incorporation for 2-3 weeks. This would have synchronised autumn N release and crop establishment better and reduced the risk of leaching losses during the 'window' between incorporation and active crop N uptake. MAFF (1991c), for example, strongly recommended delaying the autumn cultivation of soils containing high nitrogen residues as late as possible in order to delay the build-up of NO_3-N levels in the soil.

The main limitation to crop growth, however, remained the poor utilisation of legume N in spring due to its low release and limited availability under local conditions. With fertiliser N recoveries at Glencorse only in the range of 25-32%, it was unrealistic to have expected good recoveries of legume N.

Nonetheless, the forage peas still fixed large amounts of N_2 which could have been productively used within the whole farm N cycle. For example, a more appropriate way of utilising the pea's N_2 fixation potential would have been via livestock. The forage peas would have made very high quality silage (Whytock and Frame 1985) producing good financial returns in a dairy, beef or sheep enterprise, whilst still contributing to soil fertility in the arable rotation with N rich crop residues (*i.e.* over 40 kg fixed N ha^{-1} remained in the roots, Table 5.5) and the ultimate return of legume-derived N in animal manures.

6.3 SOIL N BALANCES

N balances have been constructed for a great many agricultural systems and are a useful technique for developing an understanding of the soil N cycle.

There are two general methods employed in the construction of N balances. One involves the total N balance of a system and documents all appropriate N inputs and outputs as fully as possible (*e.g.* Paustian *et al.* 1990). The other introduces a ^{15}N input, usually fertiliser, and calculates a balance based upon ^{15}N recovery (*e.g.* Dowdell and Webster 1984). Legg and Meisinger (1982) stressed that these two approaches are not equal; ^{15}N balances do not quantify total N fluxes in the system, but merely indicate how the labelled input *interacts* with the system. This is not to say that the two approaches are incompatible and ^{15}N balances may be used to indirectly estimate specific N fluxes for inclusion in a total N balance. For example, Powlson *et al.* (1986) assumed steady-state conditions in the Broadbalk Wheat Experiment at Rothamsted and used ^{15}N data from micro-plots to estimate non-fertiliser N input *i.e.* non-symbiotic fixation and dry deposition of N gases.

Once a system has been defined in space and time, its N balance is founded upon general mass balance principles, namely:

$$\text{N inputs} - \text{N outputs} = \text{Change in N storage} \qquad (27)$$

This equation has been expressed in a great number of different ways, but assuming that the system boundary is drawn just below the root zone and a 'whole crop' viewpoint is adopted, the following general form applies for a total N balance (adapted from Meisinger 1984):

$$\left[N_p + N_f + N_k + N_{cr} \right] - \left[(N_{hc} + N_{rt}) + N_g + N_l \right] = \Delta N_s + \Delta N_{om} \qquad (28)$$

where: N_p = rainfall N input,
N_f = fertiliser N input,
N_k = seed N input,
N_{cr} = N in residues from previous crop,
N_{hc} = N in harvested fraction of crop,

N_{rt} = N remaining in crop roots
N_g = gaseous N loss,
N_l = leaching loss,
ΔN_s = change in soil mineral N pool,
ΔN_{om} = change in soil organic N pool.

One of the main uses of N balances is to estimate net gains/losses of soil N status *i.e.* $\Delta N_s + \Delta N_{om}$. With simple re-arrangement it can also be used to calculate the net mineralisation (N_{min}) of soil organic matter and crop residues:

$$N_{cr} - \Delta N_{om} = \left[N_l + N_{hc} + N_{rt} + N_g + \Delta N_s \right] - \left[N_p + N_f + N_k \right] = N_{min} \qquad (29)$$

For example, equation 14 that was used to estimate N mineralisation in the zero and legume N plots by 'N recovery' (4.6.6) was effectively derived from equation 29 using the following simplifying assumptions:

N_f = 0 (none applied),
N_g = 0 (no viable field measurement),
N_k = N_{rt} (assumed because of the difficulty of measuring N_{rt} over time).

Therefore, for a given time period:

$$N_{cr} - \Delta N_{om} = \left[N_l + N_{hc} + \Delta N_s \right] - N_p = N_{min}$$

One of the great advantages of an N balance approach is that it provides an overview of the N dynamics under study, emphasising particularly the way in which different processes interact. As a final summary of the work at Glencorse, N balances have been prepared (after Paustian *et al.* 1990) for the duration of the main experimental period (September 1987 - April 1989) for each set of replicate plots .

The balances are summarised in Table 6.3 and are translated to diagramatic form in Figures 6.2 - 6.5. The nomenclature used is consistent with equation 28 and is detailed further in Figure 6.1.

Table 6.3: N balances (kg N ha^{-1}) for the four N treatments at Glencorse Mains between 4 September 1987 and 20 April 1989

	N TREATMENT AND PLOTS:			
	Recommended (4 + 7)	Reduced (1 + 6)	Zero (2 + 5)	Legume (3 + 8)
INPUTS:				
N_p	20.9	20.9	20.9	20.9
N_f	150	75	0	0
N_k	3.4	3.4	3.4	3.4
N_{cr}	15.2	6.0	16.8	358.3
Σ Inputs	: 189.5	105.3	41.1	382.6
OUTPUTS:				
N_{hc}	76.1	38.5	15.1	28.2
N_{rt}	20.5	10.4	4.1	7.6
N_g	63.7	32.9	0	0
N_l	36.0	25.8	24.6	38.9
Σ Outputs	: 196.3	107.6	43.8	74.7
ΔN_s	2.6	-12.6	-14.5	-38.8
ΔN_{om}	: -9.4	10.3	11.8	346.7
(N_{min}	: 24.6	-4.3	5.0	11.6)

Notes and assumptions:

1) N_p, N_f, N_k, N_{hc}, N_l *and* ΔN_s *were all direct measurements;*

2) N_{cr} = *root N remaining after spring barley harvest (assumed to be 27% of above-ground N yield), except for legume N treatment where:* N_{cr} = *total N content of green manure;*

3) $N_{rt} = 0.27\ N_{hc}$;

4) *All* N_{cr} *had decomposed to organic N and mineral N by the end of the experimental period, but* N_{rt} *did not start decomposing until after experiment finished (to simplify changes in residue N pool);*

5) N_g *is derived from unaccounted for* ^{15}N *in plot 7 (see 6.1.3) and assumes no* ^{15}N *uptake by biomass, but recovery of* ^{15}N *by crop roots occurs as estimated;*

6) $N_{min} = N_{cr} - \Delta N_{om}$
(values of N_{min} *included in Table 5.16 are lower because they do not include estimate of* N_{rt}*).*

Figure 6.1: N fluxes used in the estimation of N balances for the main experimental period at Glencorse Mains

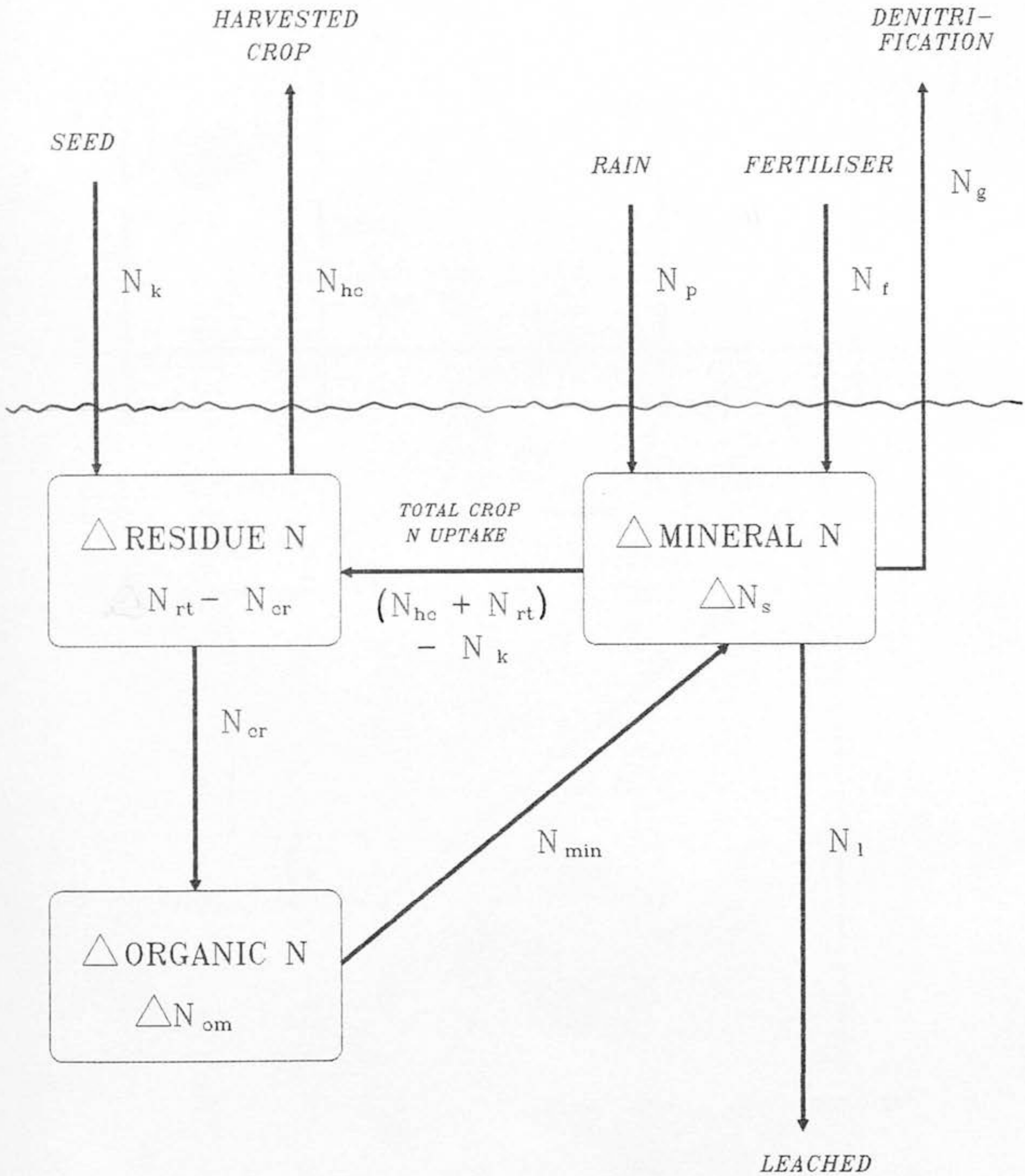


Figure 6.2: Estimated N balance for Plots 4 and 7 receiving recommended fertiliser N application (4 September, 1987 to 20 April, 1989)

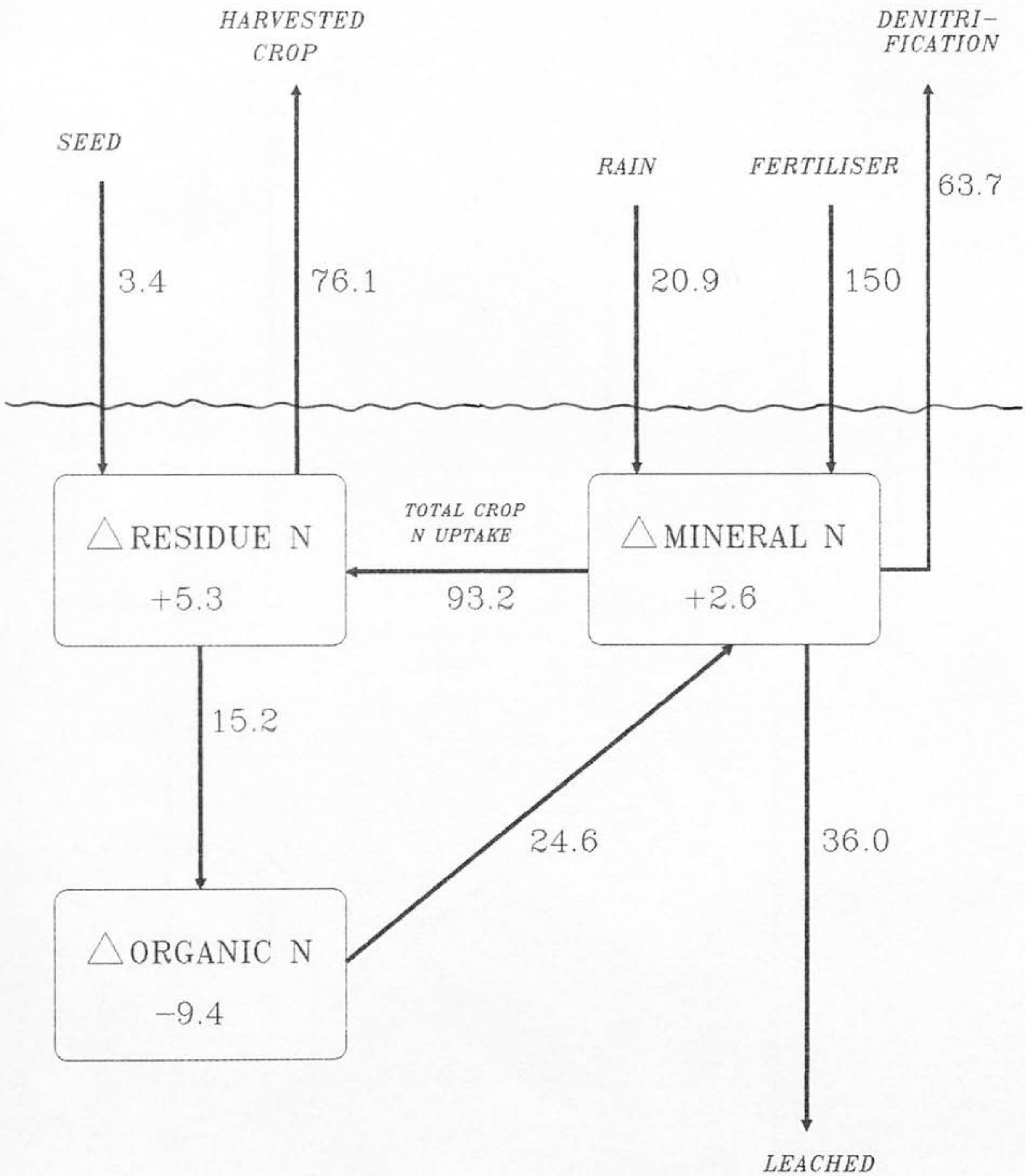


Figure 6.3: Estimated N balance for Plots 1 and 6 receiving reduced fertiliser N application (4 September, 1987 to 20 April, 1989)

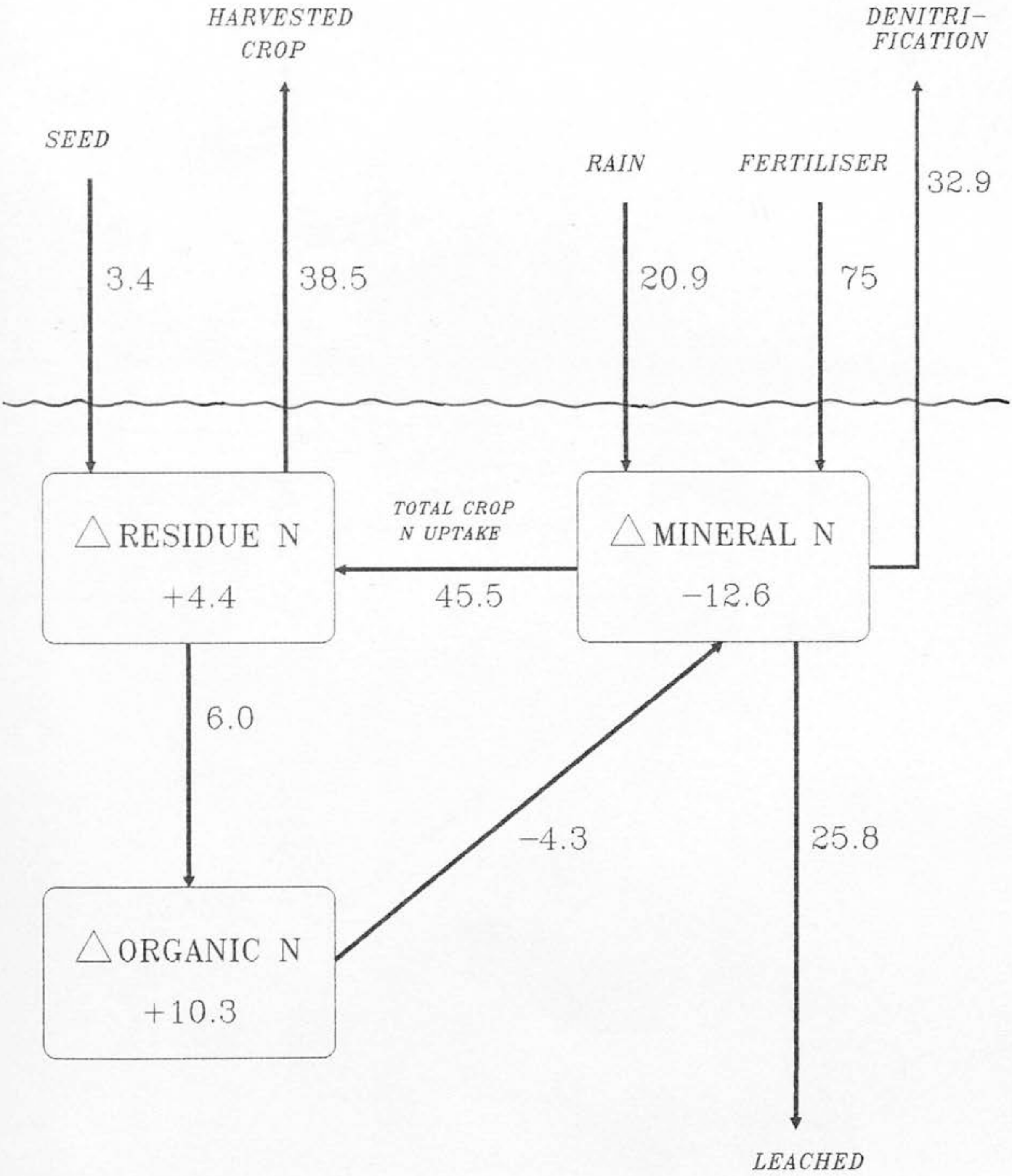
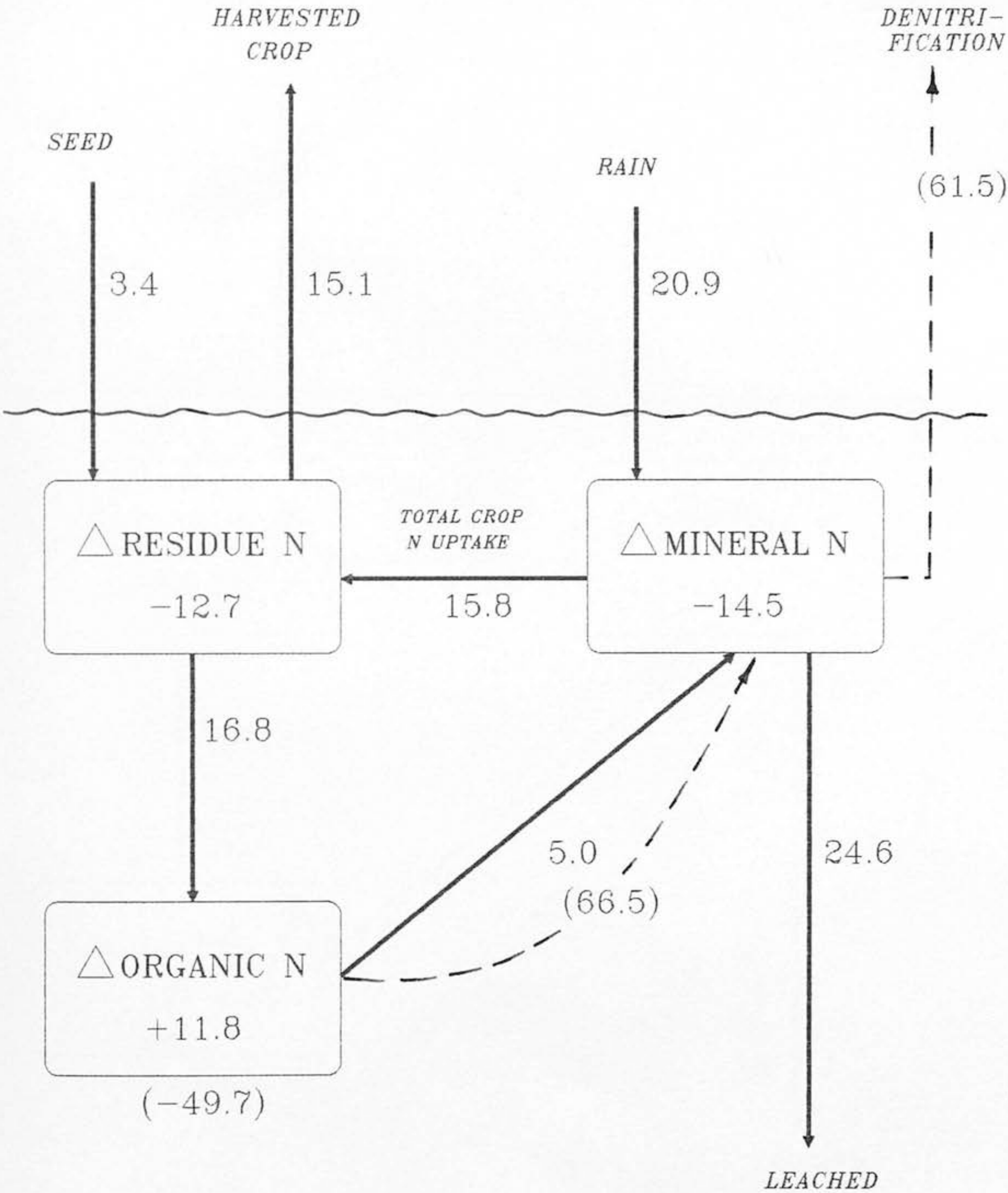
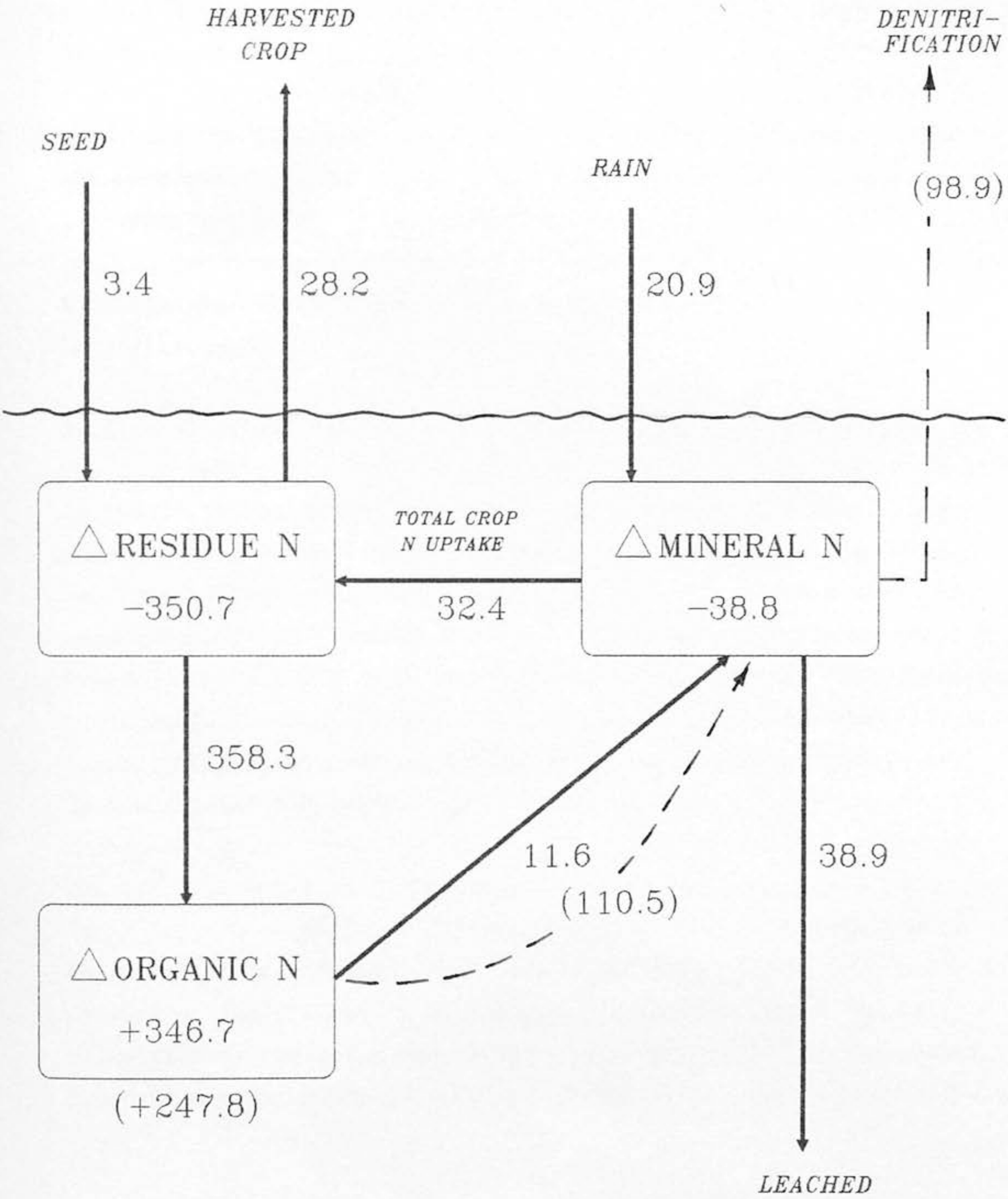


Figure 6.4: Estimated N balance for Plots 2 and 5 receiving zero N N application (4 September, 1987 to 20 April, 1989)



() = Modified fluxes assuming mineralisation data from incubated soil cores is correct

Figure 6.5: Estimated N balance for Plots 3 and 8 receiving legume N application (4 September, 1987 to 20 April, 1989)



() = Modified fluxes assuming mineralisation data from incubated soil cores is correct

Figures 6.4 and 6.5 also include the independent estimates of N_{min} derived from the field incubated soil cores. These are considerably higher than values of N_{min} calculated by equation 29 and in order for the equation still to balance the additional N supply must be matched by an increased gaseous loss (N_g).

Inclusion of a gaseous loss component gives a more realistic assessment of the N balance within the zero and legume N plots (provided the limitations of the core incubation technique are still acknowledged, 6.2.2), since it was apparent from the N_2O emissions measured that denitrification was occurring in these treatments. The problem was that, unlike the use of the ^{15}N balance in the fertiliser N treatments, there was no way of indirectly estimating the size of these losses.

Additionally, within the fertiliser N treatments it was assumed that the only denitrification losses occurring were from applied fertiliser (6.1.3). If losses were also occurring from soil-derived N (*e.g.* approximately 60 kg N ha^{-1} as suggested in Figure 6.4), then total denitrification losses from plots receiving fertiliser N were very substantial (≈ 95 - 125 kg N ha^{-1}). Whilst such figures must be treated cautiously at this stage, they do suggest that the Glencorse plots could be used for more detailed investigation of denitrification losses through the combination of several independent techniques within a total N balance.

Rosswall and Paustian (1984) formulated the concept of a N loss index for the comparison of N balances and this is of some use in the final interpretation of the Glencorse data. The N loss index is a measure of the efficiency with which a system retains N and is calculated as the total non-harvest N losses divided by the sum of external mineral N inputs plus the estimated net mineralisation, *i.e.*:

$$\text{N loss index} = \frac{N_l + N_g}{N_p + N_f + N_{min}} \times 100\% \qquad (29)$$

External mineral N input plus mineralisation can be considered as a measure of the potential availability of N to loss pathways; as more of the potentially available N is lost, so the N loss index increases.

N loss indexes for the four N treatments are given in Table 6.4. They show that, although non-harvest N losses were highest from the recommended N treatment, this treatment also had the lowest N loss index and therefore displayed the greatest ability to retain mineral N within the arable soil-plant system.

As fertiliser N application was decreased, so did non-harvest N loss, but the N loss index increased. This upward trend in N loss index may have been related to the observed positive 'priming effect' of fertiliser application on soil N uptake (6.1.2). It seems reasonable to suggest again, that the application of fertiliser N to the Winton soil encouraged more efficient utilisation of non-fertiliser N inputs by the crop (soil and/or rain N) and effectively reduced their availability for loss.

The highest N loss index was that of the legume N treatment. This did not fit the trend observed in the other treatments *i.e.* an increase in loss index also corresponded with increased non-harvest N losses compared to the zero N treatment. This presumably reflected the problem of synchronising N release and crop demand in autumn 1987.

Furthermore (although the absolute index values must be treated cautiously), an index value of over 100% further suggested that the incorporation of the green manure increased the susceptibility of non-fertiliser N inputs to loss. As already discussed (6.2.2) this 'loss' may have occurred as enhanced denitrification or microbial immobilisation and corresponds with the negative 'priming effect' observed in the ^{15}N data on legume N uptake (5.3.3). Since these processes were not quantified they do not appear explicitly as non-harvest losses, but rather exert their influence upon the N loss index by decreasing the estimated value of N_{min}.

It is interesting to note that the N loss indexes are only slightly affected when the larger mineralisation estimates are included for the zero and legume N treatments. Since all of the extra available N is presumed to be lost, the index stays reasonably constant. Therefore, in systems where there is doubt over the absolute value of certain processes, but a reasonable N balance can still be constructed, the N loss index may be a useful means of comparing different systems.

In conclusion, the recommended N treatment obviously produced the most efficient N balance in terms of minimising N losses relative to N inputs, but this must be reconciled with the fact that absolute losses were highest. The significance of this in environmental terms is not clear. The higher losses appeared to be related mainly to increased denitrification rather than leaching. If, under the extreme reducing conditions of the Winton soil, the primary product was N_2 then there is no cause for concern. But if large quantities of N_2O were being lost, these losses need to be monitored as part of the on-going concern over the contribution of intensively-farmed soils to atmospheric pollution.

Table 6.4: N loss indexes for the four N treatments during the main experimental period

Treatment:	N_l+N_g (kg N ha^{-1})	$N_p+N_f+N_{min}$ (kg N ha^{-1})	N loss index (%)
Recommended N	99.7	195.5	51.0
Reduced N	58.7	91.6	64.1
Zero N	24.6 (86.1)	25.9 (87.4)	95.0 (98.5)
Legume N	38.9 (137.8)	32.5 (131.4)	119.7 (104.9)

() = calculated including N_{min} estimates from field incubated soil cores

6.4 CONCLUSIONS

The experimental plots at Glencorse Mains occupied a marginal site for cereal production. Yields were low and the utilisation of applied N was poor. Nonetheless it was apparent that profitable cereal crops could be produced with recommended rates of fertiliser N application and there was no financial incentive to reduce these rates.

NO_3-N leaching data, although limited by variability in drainflow recovery, further suggested that there was no environmental incentive to reduce fertiliser N rates. The mineralisation of soil organic matter was probably a more significant source of NO_3-N for leaching and there was scope for reducing this through the use of winter cover crops, avoidance of autumn cultivation and improved utilisation of late season mineralisation by spring cereals. The application of higher rates of fertiliser N also appeared to improve the cereal crops' utilisation of soil-derived N.

However, before concluding that fertiliser use at the site posed no significant environmental risk at all, more information is needed about N_2O emissions particularly from spring applied N.

The site was clearly capable of supporting very high levels of symbiotic N_2 fixation, but the utilisation of this via the growth and incorporation of a leguminous green manure was not successful. The dynamics of the legume N input were considerably more complex than those of the fertiliser N inputs and led to the poor synchronisation of N supply and crop N demand. This significantly increased leaching losses in the autumn, but severely restricted crop growth in the spring. The only agronomic value that could be attributed to the green manure was a possible increase in soil N status.

There would be considerably more potential for exploiting such high rates of N_2 fixation within a mixed rotation, incorporating livestock enterprises, rather than a strictly arable rotation.

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8 Nitrogen Dynamics of a Leguminous Green Manure

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ABSTRACT

The effective use of leguminous crop residues as a source of nitrogen (N) for subsequent crops requires an understanding of the soil N dynamics following incorporation of the legume material. This study investigated the short-term N dynamics of an arable soil in south-east Scotland between the autumn incorporation of a field-grown green manure crop (forage peas) and harvesting of the subsequent cereal crop (winter barley).

Direct in-field measurements were made of nitrate leaching losses, crop N uptake, N_2O emissions and soil mineral N levels. N release from the green manure was assessed by two independent estimates of N mineralization.

The total N input provided by the green manure was 335 kg N ha^{-1}, but the 'apparent' recovery of legume N (calculated by difference) in the following winter barley crop was only 13.3 kg N ha^{-1}. This low N uptake was apparently due to the poor release of mineral N from the legume residues, although denitrification losses may also have reduced the NO_3-N available for crop uptake. Leaching losses from the green manure were also low, and should present little environmental risk.

This work suggested that the use of green manures to increase available N for following crops may be limited under the soil and climatic conditions at this site.

INTRODUCTION

The growth and incorporation of leguminous crops is a potentially useful means of exploiting biological N_2 fixation as an additional/alternative N input in the arable rotation. Optimum use of such legume N for subsequent crop growth requires an understanding of the complex soil N dynamics following incorporation of the legume material.

In the UK, Dyke *et al.* (1977) reported considerable cereal yield increases after the growth of trefoil as a leguminous green manure. Groffman *et al.* (1987) also suggested that legume N inputs from winter cover crops in North America have the potential to produce large amounts of available N for crop uptake with minimal environmental risk. Meanwhile in Australia, Ladd *et al.* (1981b) reported that the majority of ^{15}N applied to soils as labelled legume material was retained as soil organic N, with relatively little ^{15}N taken up by the crop. They concluded that the value of decomposing legume material was to maintain long-term soil organic N status, rather than to supply large amounts of mineral N for short-term crop uptake. Müller and Sundman (1988) reached a similar conclusion in Finland, using a variety of ^{15}N-labelled legume material. Müller (1987) also reported low leaching losses from buried clover material in lysimeters. However, this is contrary to the high nitrate leaching losses reported by Adams and Pattinson (1985) following the incorporation of clover residues in a legume-based crop rotation in New Zealand.

The object of this study was to investigate the short-term N dynamics of an arable soil following the growth and incorporation of a leguminous green manure under the field conditions prevalent in south-east Scotland. This involved the direct, in-field measurement of nitrate leaching losses, crop uptake, nitrous oxide emission and soil mineral N levels between incorporation of the green manure and harvesting of the subsequent cereal crop. Two independent methods were used to estimate N mineralization.

MATERIALS AND METHODS

Experimental site

This work was conducted between August 1987 and July 1988 on Edinburgh University's Bush Estate, as part of a larger N balance study investigating the effect of four experimental treatments, on winter barley (*Hordeum vulgare* L.):

(1) High fertilizer N application (150 kg N ha^{-1}),

(2) Low fertilizer N application (75 kg N ha^{-1}),

(3) Zero N application,

(4) Organic N application – growth and incorporation of a leguminous green manure.

Only the results from the Zero and Organic N treatments are reported

Table 1. Soil details (plough layer only).

Soil series:	Winton
Soil association:	Rowanhill
Textural class:	clay loam
Organic matter (%):	4.8–6.0
pH:	6.2–6.7
Clay (%):	24
US classification:	Typic haplaquept

here, whilst full details of the larger N balance study will be published elsewhere.

The basis of the larger N balance experiment was the direct, in-field measurement of nitrate leaching losses from eight, 300 m^2, hydrologically isolated field plots. The plots were established in April 1987, as two replicate blocks on a glacial-till-derived soil of the Winton series. Soil details are given in Table 1. The Winton series typically displays heavily compacted and imperfectly drained B and C horizons. Therefore the plots were established on a sloping (5%) site such that plot isolation could be achieved by the interception and collection of downslope lateral flow in a deep ditch. A pipe led from each plot collection ditch to an instrument pit where drainflow was monitored by a tipping bucket flow meter (Field Drainage Experimental Unit, Cambridge, UK) connected to a data logging system (Cristie Electronics Ltd, Gloucestershire, UK). Drainflow was continuously and proportionately sampled by a simple device which collected about 10 ml of the drainage water dispensed from every second tip of the tipping buckets. The 10 ml samples were accumulated and a single bulk sample was collected weekly for analysis. Nitrate N (NO_3-N) and ammonium N (NH_4-N) levels in the water samples were determined by continuous-flow analysis, using the methods of Henricksen and Selmer-Olsen (1970) and Crooke and Simpson (1971) respectively. NH_4-N levels were insignificant and analysis was not continued.

Crop husbandry

Before the establishment of the winter barley, the Zero N treatment was spring barley + 120 kg N ha^{-1} and the Organic N treatment was a green manure crop of forage peas (*Pisum arvense* L. cv. Birte). The peas were sown in April 1987 at a seed rate of 227 kg ha^{-1}, mixed with 25 kg ha^{-1} of spring barley. This intercropping was to prevent the peas lodging. 20 kg N ha^{-1} was applied to the peas at emergence to encourage rapid establishment.

The peas were chopped by rotary-mower on 19 August 1987 at the full bloom/flat pod growth stage after 16 weeks growth. To determine the N content of the pea herbage and roots, plant samples were harvested from 3.0 m^2 and two 20 × 20 × 25 cm soil cores were taken from each pea plot before chopping. Roots were extracted from the soil cores using a wet sieving procedure. Herbage and root material was dried for 24 hours at 105°C, milled, and duplicate samples were analysed for total N content with a Carlo-Erba 1400 automatic N analyser.

In the field the chopped pea material was left on the soil surface until 4 September 1987, then incorporated into the soil using a rotary-cultivator. The depth of incorporation was only about 10 cm to encourage an initial aerobic decomposition. On 22 September 1987 all the plots, including the Organic N treatments, were conventionally ploughed and following cultivation the winter barley (cv. Marinka) was sown at a seed rate of 210 kg ha^{-1}. Basal P and K dressings were applied at this time. The winter barley was harvested at maturity on 26 July 1988.

Leaching losses

Routine weekly leaching measurements began in September 1987, when the soil returned to field capacity, and were prepared as monthly cumulative plot values. The apparent recovery of incident rainfall as drainage from the plots was variable, as indicated in Table 2, and so some correction of leaching losses was required, as follows.

Plot 2 (Zero N) was susceptible to surface run-on from surrounding areas during heavy rainfall, but it was assumed that the run-on contained little N and no correction was made for this.

Plots 3 and 8 (Organic N) showed low recoveries of incident rainfall (Table 2), and measureable drainflow only occurred following relatively large storm events. Subsequent detailed investigation of the relationship between drainflow rate and NO_3-N concentration (using spot-samples taken on an 8-hour cycle) suggested no significant bias in the estimation of average NO_3-N concentration if only high flow rates were sampled. Therefore it was assumed

Table 2. Apparent recovery of incident rainfall (%) from the Zero and Organic N plots during the experimental period (19 August 1987-26 July 1988).

Plot number	Treatment	Apparent recovery of incident rainfall (%)
2	Zero N	67
3	Organic N	6
5	Zero N	67
8	Organic N	24

an unbiased estimate of leaching losses from plots 3 and 8 could simply be made by multiplying the measured monthly NO_3-N loss by the ratio of expected/actual monthly rainfall recovery. The expected recovery was taken as the observed monthly rainfall recovery from the adjacent plot 7.

Denitrification losses

Denitrification losses were monitored weekly from the end of September 1987, using an adaptation of the acetylene-inhibition technique described by Ryden *et al.* (1979). In each plot one 0.5 m^2 sealed steel canopy was flooded with acetylene and left for 24 hours before sampling for N_2O. It is unlikely that acetylene inhibition was particularly successful on the Winton soil series (Arah, 1988) and no attempt was made to quantify denitrification losses. Instead the 24-hour N_2O emissions are reported simply for qualitative comparison between the treatments, and to identify periods during which denitrification may have caused errors in the estimation of mineralization.

Crop uptake and soil mineral N

Crop N uptake and soil mineral N levels were measured regularly from two microplot areas marked within each plot. Two 1 m rows (0.24 m^2 area) of barley were harvested from each microplot and analysed for total N content as for the pea material. At final harvest an additional twenty 1 m rows (2.40 m^2 area) were cut from a diagonal transect across the plots for a more accurate estimate of total dry matter yield and N uptake.

At each sampling date four 20 cm deep soil cores were taken from each microplot. The cores for each microplot were bulked, sieved and duplicate samples were extracted by shaking for 1 hour with 1 M KCl. Extracts were analysed for NO_3-N and NH_4-N by the same procedure as for the water samples and, following correction for soil moisture content and bulk density, were expressed both as mg N kg^{-1} soil and kg N ha^{-1}. Crop N uptake and soil mineral N data were prepared as mean plot values for each sampling date.

Field incubation of soil cores

Starting from 20 October 1987 and at each subsequent sampling date, another four 20 cm soil cores were taken from each microplot, placed as intact as possible into plastic bags and returned to their holes. A small perforation was made in each bag to limit the development of anaerobic conditions, and a plastic cover was placed over each core to prevent water entering. Following field incubation, cores from each microplot were removed, bulked and extracted for NH_4-N and NO_3-N as above. After correction for soil moisture content and bulk density the results were expressed as kg N ha^{-1} and prepared as a mean value for each plot.

No Zero N incubation was made between March and April 1988, but it has been assumed that the rate of accumulation of NH_4-N and NO_3-N in the Zero N plots between March and April was the same as between January and March 1988.

Estimation of N mineralization

N mineralization in both treatments was estimated by two approaches.

Method 1

N mineralization was estimated for each plot from the 'recovery' of N in leaching losses, crop N content and soil mineral pools at 0, 62, 97, 161, 211, 245, 273, 307 and 342 days after chopping the peas (Fig. 5). Total N mineralization, TN_{min} (kg N ha^{-1}), since chopping the peas, was calculated as follows:

$$TN_{min}(t) = TN_{l(t)} + TN_{c(t)} + [N_{(t)} - N_{(0)}] \quad (1)$$

where $TN_{l(t)}$ = cumulative N leaching losses (kg N ha^{-1}) at time t, $TN_{(t)}$ = crop N content (kg N ha^{-1}) at time t, $N_{(t)}$ = soil mineral N content (kg N ha^{-1}) at time t; $N_{(0)}$ = soil mineral N content (kg N ha^{-1}) at day 0.

Net mineralization, N_{min} (kg N ha^{-1}), for a given time period ($t-1$ to t) was estimated by

$$N_{min(t)} = TN_{min(t)} - TN_{min(t-1)} \quad (2)$$

Method 2

The use of field-incubated soil cores gave an independent estimate of N mineralization (Fig. 6) for comparison with method 1. Field incubations began 62 days after the chopping of the peas and continued sequentially to 97, 161, 211, 245, 273, 307 and 342 days after chopping. Net mineralization, N^*_{min} (kg N ha^{-1}), for each plot and incubation period was estimated using a similar rationale to that proposed by Raison *et al.* (1987) as follows:

$$N^*_{min(t)} = N_{b(t)} - N_{(t-1)} \quad (3)$$

where $N_{b(t)}$ = mean mineral N content (kg N ha^{-1}) of the soil cores in the plastic bags at the end of incubation, time t; $N_{(t-1)}$ = soil mineral N content (kg N ha^{-1}) measured at the start of incubation, time $t-1$.

Total N mineralization, $TN^{\bullet}_{min}$ (kg N ha^{-1}), from the start of incubations to time t, was calculated as the sum of $N^{\bullet}_{min}$ values for all preceding periods.

'Apparent' mineralization of legume material

The net 'apparent' mineralization of legume material for a given time period was derived from Equations (2) and (3) respectively as follows:

$$L_{min(t)} = N^{O}_{min(t)} - N^{Z}_{min(t)} \quad (4)$$

$$L^{\bullet}_{min(t)} = N^{\bullet O}_{min(t)} - N^{\bullet Z}_{min(t)} \quad (5)$$

where the superscripts O and Z refer to the Organic and Zero N treatments respectively.

Statistical analysis

Treatment means were prepared from the plot means. Standard errors were estimated for each treatment mean using a 'stabilized' estimate of the error mean square for all sample points. This method was used because of the low level of plot replication, and gives a conservative estimate of error (Hunter, personal communication). Confidence intervals of 95% were calculated (using 't-values') for all sample points as a test of the significance of treatment mean differences.

RESULTS AND DISCUSSION

The total N input of the forage peas when incorporated into the soil was approximately 335 kg N ha^{-1}. On a dry matter basis, this input consisted of 9.6 t ha^{-1} herbage material at 3.1% N content and 2.2 t ha^{-1} root material at 1.4% N. Thus the material incorporated was mainly of low C/N ratio.

Crop uptake

Total crop N uptake from the Zero and Organic N treatments was 15.0 and 28.3 kg N ha^{-1} respectively (Fig. 1, Table 3). The Zero N uptake was low compared with other field-trial results on similar soil types in south-east Scotland. For example, Smith *et al.* (1984) obtained crop uptakes of 18–30 kg

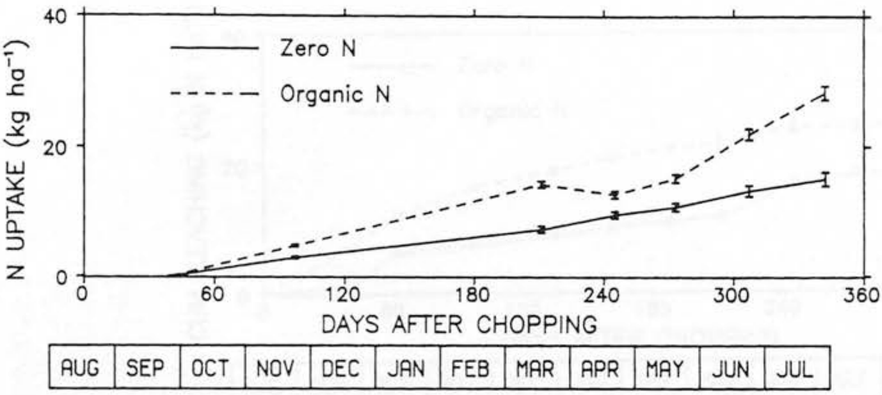

Fig. 1. N uptake (kg N ha^{-1}) of winter barley after drilling on 28 September 1987 (bars are standard errors).

Table 3. Estimated values of the total net 'apparent' mineralization (kg N ha^{-1}) of incorporated legume material from method 1 (N recovery) and method 2 (soil incubation).

Method 1: Aug. 1987 – July 1988	Crop N uptake	Leaching losses	Change in soil mineral N	TN_{min}
Zero N:	15.0	20.1	−1.2	33.9
Organic N:	28.3	26.2	0.6	55.1
			L_{min}:	21.2

Method 2: Oct. 1987 – July 1988	TN^*_{min}	$(TN_{min})^*$
Zero N:	83.4	(45.0)
Organic N:	101.4	(54.4)
L^*_{min}:	18.0	$(L_{min})^*$: 9.4)

* Estimation of L_{min} by method 1 for the period October 1987 to July 1988 in parentheses.

N ha^{-1} from Zero N plots of spring barley on a Winton/Macmerry series soil. The Organic N uptake was consistently greater than the Zero N during the season, and at harvest the 'apparent' uptake (calculated by difference) of legume-derived N by the barley was 13.3 (s.e. = 1.5) kg N ha^{-1}. This only represented a 4% recovery of incorporated legume N by the winter barley. Other workers have also found relatively low crop recoveries of legume N, for

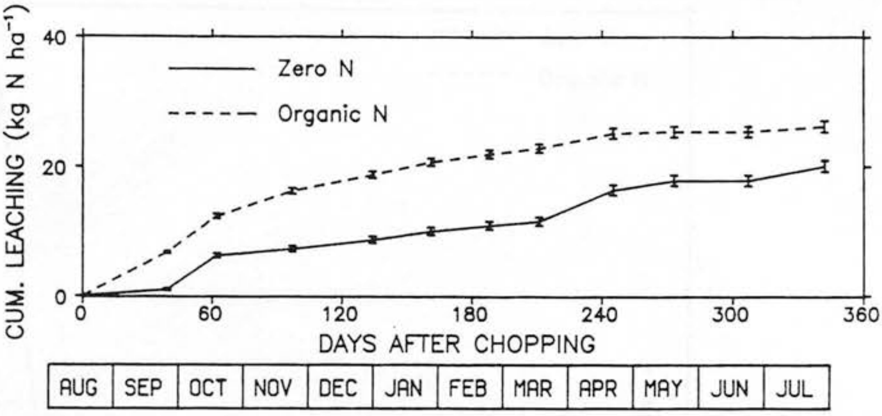

Fig. 2. Cumulative NO_3-N leaching losses (kg N ha^{-1}) after chopping peas on 19 August 1987 (bars are standard errors).

example Ladd *et al.* (1981b) reported only 10.9–17.3% of applied ^{15}N-labelled legume N taken up by wheat plants, whilst 71.9–77.7% remained as soil organic N.

Leaching losses

Total leaching losses from the Zero and Organic N treatments were 20.1 and 26.2 kg N ha^{-1} respectively (Fig. 2, Table 3). Total leaching losses from the Organic N treatment were low compared to other work. Adams and Pattinson (1985) reported leaching losses of approximately 90 kg N ha^{-1} after the incorporation of a white clover ley and 60 kg N ha^{-1} after incorporating arable pea residues. Also Bergström (1987) measured leaching losses of up to 42 kg N ha^{-1} from lysimeters and field plots during the first 20 weeks after incorporation of a grass ley. The 'apparent' leaching loss of legume-derived N (calculated by difference) during the experimental period was 6.1 (s.e. = 1.3) kg N ha^{-1}, which was less than 2% of the legume N input. A similar low leaching loss of legume-derived N was reported by Müller (1987) using ^{15}N-labelled clover buried in lysimeters.

The initial shallow incorporation of the pea material into the soil in August increased soil NO_3-N levels (Fig. 3) and produced significantly higher leaching losses (Fig. 2) from the Organic N treatment during September, indicating a rapid initial decomposition of the fresh legume material between incorporation and the establishment of the winter barley. This illustrates the problem of synchronizing legume N release and crop N requirement during the early autumn which was probably accentuated in this experiment by specifically encouraging aerobic soil conditions after legume incorporation.

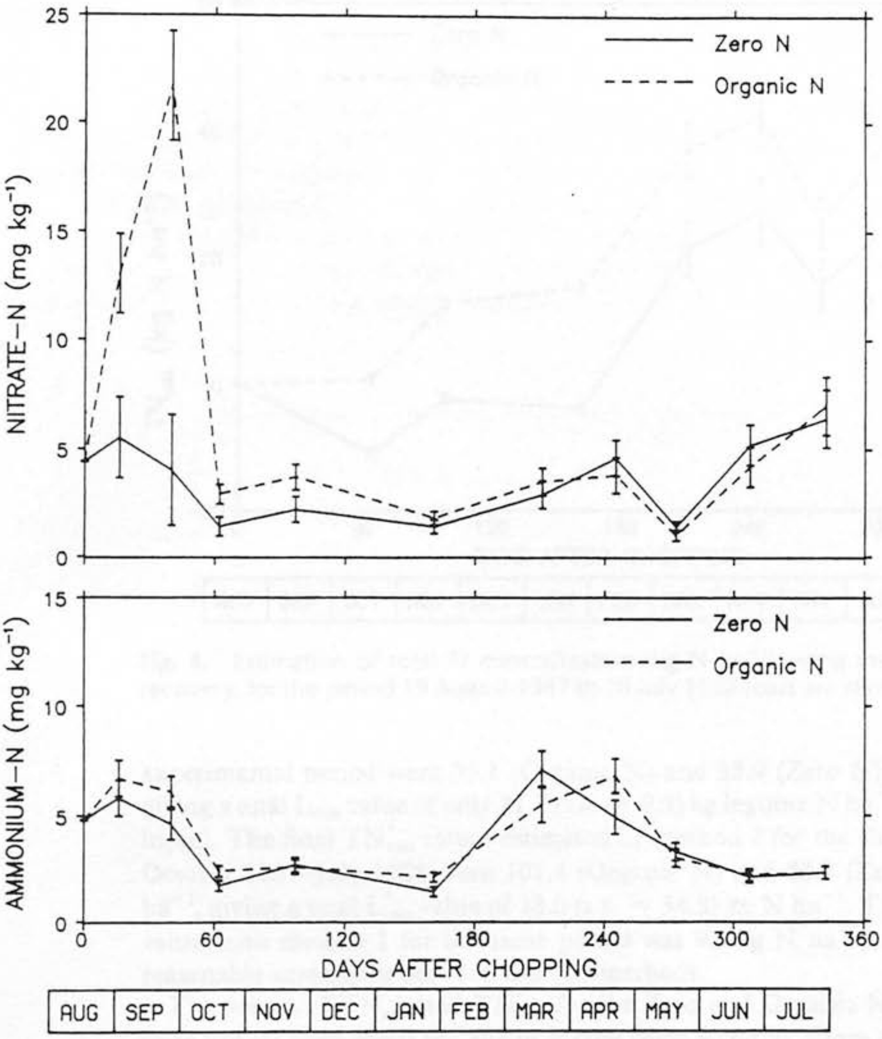

Fig. 3. Soil NH_4-N and NO_3-N levels (mg N kg^{-1} soil) in 0–20 cm layer after chopping peas on 19 August 1987 (bars are standard errors).

N mineralization

During the experimental period there was no significant difference in the pattern of N mineralization from the Zero and Organic N treatments, when estimated by method 1 (N recovery – Fig. 4) and method 2 (soil incubation – Fig. 5). The final values of TN_{min} estimated by method 1 for the whole

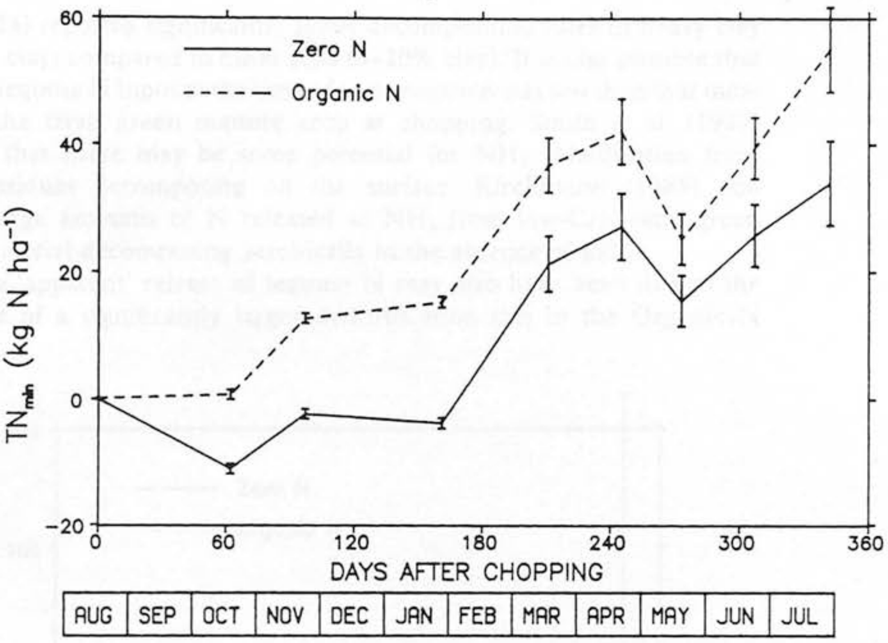

Fig. 4. Estimation of total N mineralization (kg N ha^{-1}) using method 1 – N recovery, for the period 19 August 1987 to 26 July 1988 (bars are standard errors).

experimental period were 55.1 (Organic N) and 33.9 (Zero N) kg N ha^{-1}, giving a total L_{min} value of only 21.2 (s.e. = 9.5) kg legume N ha^{-1} (6% of the input). The final TN^{*}_{min} values estimated by method 2 for the shorter period October 1987–July 1988 were 101.4 (Organic N) and 83.4 (Zero N) kg N ha^{-1}, giving a total L^{*}_{min} value of 18.0 (s.e. = 34.8) kg N ha^{-1}. The total L_{min} value from method 1 for the same period was 9.4 kg N ha^{-1}, suggesting a reasonable agreement between the two methods.

The values of TN_{min} and TN^{*}_{min} for the Zero and Organic N treatments were similar until about the end of March (Figs 4 and 5). From April to July however, values of TN^{*}_{min} increased far more rapidly than the values of TN_{min}, suggesting enhanced mineralization in the contained soil cores. Reasons for such an enhancement are not clear, but might be due to the maintenance of a favourable soil moisture regime within the plastic bags whilst the bulk of the soil in the plots was drying out.

A number of factors may have contributed to the low 'apparent' release of legume N observed in this work. Low soil temperatures would have reduced the decomposition rate of the added legume material (for example Ladd *et al.*, 1985) and poor soil physical conditions may also have had some effect. Ladd *et al.* (1981a) reported significantly lower decomposition rates in heavy clay soils (42% clay) compared to other soils (5–20% clay). It is also possible that the actual legume N input at the time of incorporation was less than that measured in the fresh green manure crop at chopping. Smith *et al.* (1987) suggested that there may be some potential for NH_3 volatilization from legume residues decomposing on the surface. Kirchmann (1985) also reported large amounts of N released as NH_3 from low-C/N-ratio green manure material decomposing aerobically in the absence of soil.

The low 'apparent' release of legume N may also have been due to the occurrence of a significantly larger denitrification loss in the Organic N

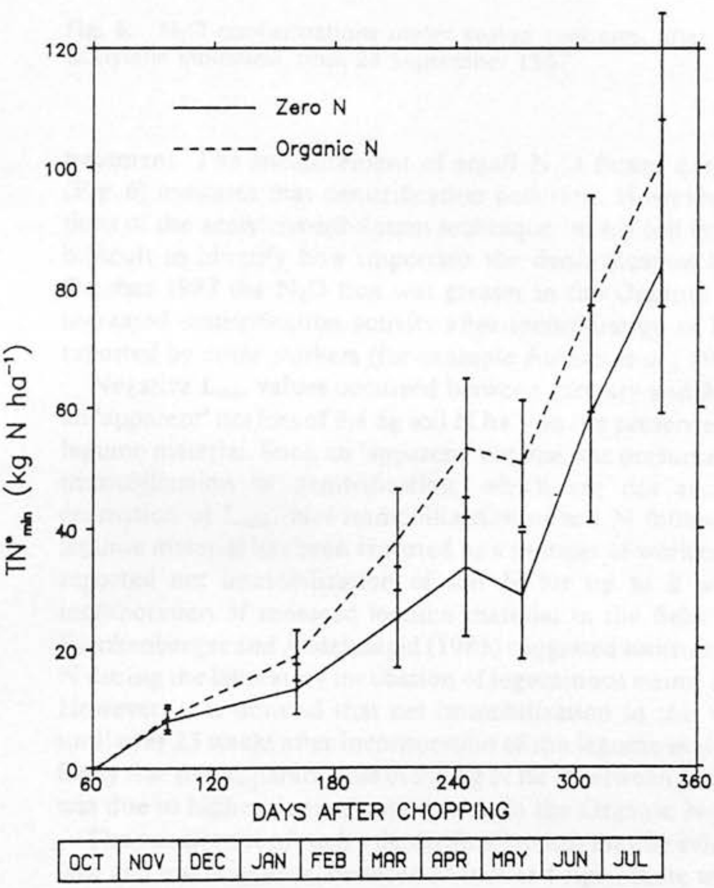

Fig. 5. Estimation of total N mineralization (kg N ha^{-1}) using method 2 – soil incubation, for the period 20 October 1987 to 26 July 1988 (bars are standard errors).

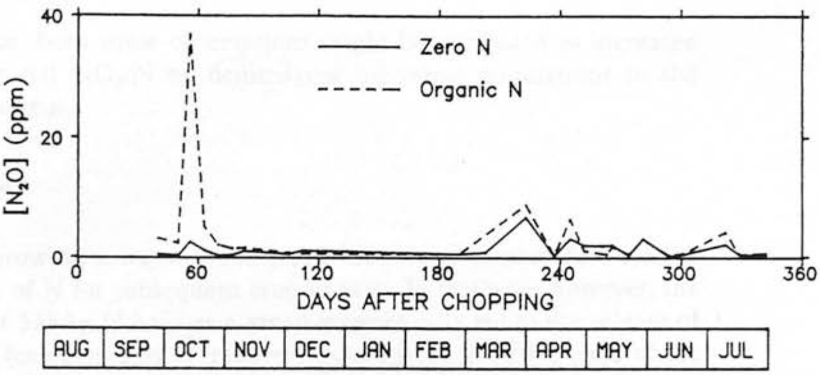

Fig. 6. N_2O concentrations under sealed canopies, after 24 hours and partial acetylene inhibition, from 24 September 1987.

treatment. The measurement of small N_2O fluxes during the experiment (Fig. 6) indicates that denitrification occurred. However, due to the limitations of the acetylene-inhibition technique in this soil type (Arah, 1988) it is difficult to identify how important the denitrification losses were. During October 1987 the N_2O flux was greater in the Organic N plots, suggesting increased denitrification activity after incorporation of legume material, as reported by other workers (for example Aulakh *et al.*, 1983).

Negative L_{min} values occurred between January and May 1988 indicating an 'apparent' net loss of 9.4 kg soil N ha^{-1} in the presence of the incorporated legume material. Such an 'apparent' net loss was presumably related to either immobilization or denitrification, which are not accounted for in the estimation of L_{min}. Net immobilization of soil N following the addition of legume material has been reported by a number of workers. Ladd *et al.* (1986) reported net immobilization of soil N for up to 8 weeks following the incorporation of senesced legume material in the field. Also the results of Frankenberger and Abdelmagid (1985) suggested some immobilization of soil N during the laboratory incubation of leguminous stems, even after 20 weeks. However, it is unusual that net immobilization in this work did not occur until over 23 weeks after incorporation of the legume material. It seems more likely that the 'apparent' loss of 9.4 kg N ha^{-1} between January and May 1988 was due to higher denitrification rates in the Organic N plots.

The occurrence of such a denitrification loss may be evident in the crop uptake and leaching data. Between March and April there was a lower leaching loss from the Organic N than Zero N treatment (Fig. 2); and a reduction in N uptake from the Organic N treatment (Fig. 1). It is unlikely that this was an actual loss of crop N, but it probably reflected at least some reduction in the rate of N uptake. Both these observations might be attributed to increased competition for soil NO_3-N by denitrifying microbial populations in the Organic N treatment.

CONCLUSIONS

The object of growing a leguminous green manure is to provide a readily available source of N for subsequent crop growth. In this work however, the incorporation of 335 kg N ha^{-1} as a green manure only led to the release of 21.2 kg N ha^{-1} (calculated by N 'recovery') during the growth period of the following crop. Of this, 13.3 kg N ha^{-1} was recovered in the crop and 6.1 kg N ha^{-1} lost by leaching. The incorporation of leguminous green manures therefore presents little environmental risk to groundwater under local conditions.

It is likely that denitrification was an important loss process after the incorporation of the green manure, resulting in a reduction in availability of both legume- and soil-derived N at certain times.

At this stage of our work it seems reasonable to reiterate the conclusions of Ladd *et al.* (1981b) and Müller and Sundman (1988), that the value of leguminous green manures appears to be the long-term maintenance of soil N status rather than the short-term supply of available N for crop growth.

ACKNOWLEDGEMENTS

The authors thank Ms C. Runciman for diligent technical assistance, E. A. Hunter of the Scottish Agricultural Statistics Service for statistical advice and the staff of the Crop Production Department of the Edinburgh School of Agriculture for assistance with field operations. The work was funded by the Department of Agriculture and Fisheries for Scotland.

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Journal of Soil Science, 1990, **41**, 199–214

Calibration and validation of a model of non-interactive solute leaching in a clay–loam arable soil

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SUMMARY

A capacity-type approximate leaching model (Addiscott *et al.*, 1986) with a simple treatment of soil matrix permeability was tested, using field tracer experiments with $CaBr_2$, on hydrologically isolated plots. The model predictions are sensitive to the value of the soil matrix permeability factor, a, and four methods of estimating this parameter were evaluated: (1) using a calibration based on soil texture; (2) least-squares fitting of the model to successive neutron probe measurements of the water content profile; (3) least-squares fitting to daily drainage outflow; (4) least-squares fitting to cumulative drainage outflow. The best independent method (method 3) led to slight (20–30%) under-prediction of leaching losses for two of the four experiments, but in one experiment leaching was much less than predicted. As a management model the approach seems promising but more attention needs to be paid to estimation of the value and variability of the permeability parameter, a.

The convective-dispersion equation, using steady-state assumptions and a fitted dispersion length, gave as good a prediction of cumulative leaching losses as the approximate model studied here.

INTRODUCTION

Conditions for arable farming in the south-east of Scotland are very variable because of the wide range of elevation, mean annual temperature and annual effective precipitation. However, over much of the area which grows cereals, the rate of mineralization of N during winter is low. This, combined with a winter excess rainfall that is higher than for much of the arable area of the UK (Chandler & Gregory, 1976, p. 197), means that rather low carry-over of mineral N to spring occurs. Therefore, it is critical, for both winter and spring cereals, that inorganic fertilizer N is applied early. However, this early application is vulnerable to leaching; thus the prediction of leaching of early-spring fertilizer N is important so that later applications can be adjusted. At present, N adjustments in Scotland are primarily based on the observed relationship between yield and fertilizer N application, with subsequent very approximate recommendations to account for soil N supply and leaching losses after fertilizer application.

Many of the arable soils of south-east Scotland are surface water gleys derived from glacial till of sandy clay loam/clay loam texture with compact subsoils. Tracer experiments with ^{15}N-labelled fertilizer indicate little early exploitation of soil N below 30 cm (Smith *et al.*, 1984) because root exploration is inhibited by this subsoil compaction. This also means that water flow in these soils in winter is chiefly a combination of vertical transport in the plough layer and lateral, saturated interflow at the base of the plough layer, especially on sloping sites. Artificial drainage, therefore, must be connected by permeable backfill to the plough layer in order to be effective. Vertical flow may either be through the soil matrix or by rapid bypass flow through structural pores and channels.

Little information exists on leaching of N from such soils—indeed it is often assumed that leaching is minimal because of the heavy texture and the imperfect drainage. However, leaching losses, particularly of fertilizer N on sloping sites, may be serious and the objective of this work was to investigate the validity of a simple capacity-type leaching model to predict this leaching (Addiscott *et al.*, 1986).

This is a management model of solute leaching incorporating a simple measure of soil permeability designed to yield practical advice with a minimum of input parameters. The model is suited to the situation where the soil water can be partitioned into mobile and immobile phases (van Genuchten & Wierenga, 1976) and is a good approximation in many heavy-textured soils and where movement of water by bypass flow down structural pores and cracks can occur when the soil surface layer reaches saturation. A similar model has recently been published by Barraclough (1989a). This uses a hydraulic conductivity *vs* water content function estimated from moisture release data rather than an empirical permeability parameter and solves the convective-dispersion equation explicitly. A critical part of both models is prediction of the point at which bypass flow begins to occur, for if fertilizer has just been applied, this bypass flow can be very effective in causing leaching.

Barraclough (1989b) obtains reasonable predictions of winter nitrate leaching with his model. Addiscott *et al.* (1978) attempted to validate an earlier version of the Addiscott (1977) model using data from the Rothamsted drain gauges. A more recent version was also tested on a range of heterogeneous field soils, and it was found necessary to account for spatial variability of model parameters to obtain the best agreement (Addiscott & Bland, 1988). In none of these cases is testing done specifically to look at the leaching of recently applied fertilizer, which is especially prone to leaching via bypass flow.

THEORY

Water storage and movement

The most appropriate management-oriented solute leaching model for the soil and site considered here was thought to be that of Addiscott *et al.* (1986). Fig. 1 illustrates the simple description of water movement used in the model. Water storage capacity is divided into three compartments in each successive soil layer:

$$\theta_e = 0.5\theta_{(\psi = -1500\,\mathrm{kPa})} \tag{1}$$

$$\theta_r = \theta_{(\psi = -33\,\mathrm{kPa})} - \theta_e \tag{2}$$

$$\theta_{ms} = \theta_{(\psi = 0\,\mathrm{kPa})} - \theta_r \tag{3}$$

where θ = volumetric water content (m^3m^{-3}),

θ_e = volumetric water content from which anions are excluded (m^3m^{-3}),

θ_r = immobile volumetric water content (m^3m^{-3})

θ_{ms} = volumetric water content at saturation (m^3m^{-3}),

ψ = soil matric potential (kPa).

These compartments correspond to the volume of water from which solute is excluded completely (θ_e), the volume of water that is immobile—i.e. present within structural aggregates and only moving extremely slowly in response to hydraulic gradients, but within which solute is free to diffuse (θ_r), and the volume of water that is mobile in the soil and which redistributes more or less quickly to establish hydraulic equilibrium (θ_{ms}). The lower boundary of the mobile water compartment is the 'field capacity' estimate often used in the USA, and corresponds to a value of ψ of -33 kPa. In heavy soils the matric potential rarely, if ever, reaches such a low value during the winter months so the upper limit of this mobile fraction is set as that corresponding to saturation (θ_{ms}). Actual mobile water content at any time is given by θ_m. A value of θ_m corresponding to a matric potential of -5 kPa is used to generate default values of soil water content where this is appropriate. Only a proportion, α, of the water present in the mobile water compartment moves on to the next compartment (see Fig. 1) the same day. Any water introduced into the first layer of soil in excess of saturation is assumed to move directly to the drainage system (mole or pipe), as bypass flow via gravitational pores, without interacting with layers below the top one.

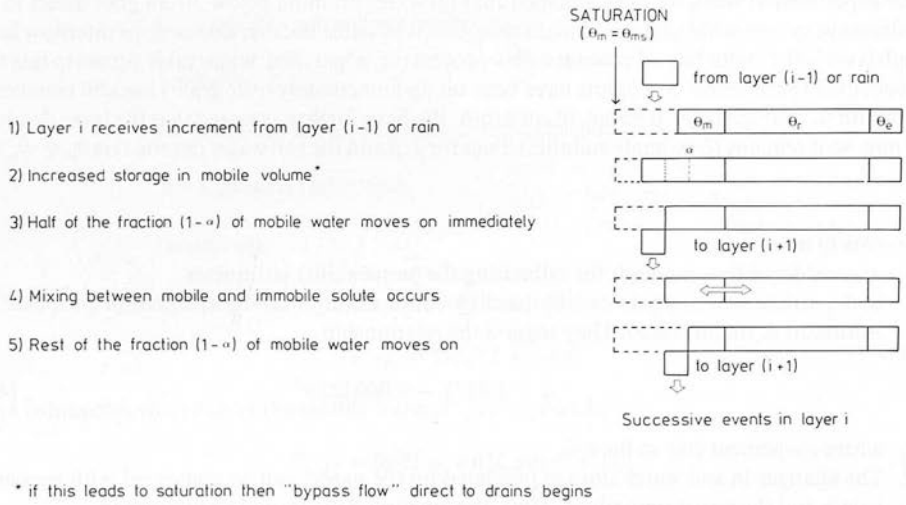

Fig. 1. Layer model of water flow and solute transport (Addiscott *et al.*, 1986).

Solute transport

Non-adsorbed solute ions can be present in mobile or immobile water. Water entering layer 1 in rainfall will equilibrate immediately with solution in the mobile water in layer 1, but not with solute in the immobile fraction. Of the fraction α of the water in the mobile fraction which moves on the first day, half (i.e. $\alpha/2$) is transported immediately and then partial equilibration between the solute

Table 1. Parameters for the layer model of Addiscott *et al.* (1986)

α	The proportion of water in mobile fraction which moves on from the original layer after 1 d.
β	The fraction of solute held in retained water which is 'held back' from equilibration with the solute in mobile water each day.
M	The soil moisture deficit from field capacity, $\theta_r + \theta_e$, at time = 0. This is set at zero if mobile water occurs in the soil layer.
L	The layer depth, set by default to 50 mm.

in immobile water and solute in the remaining mobile water occurs. The extent of this partial equilibration is described by the parameter β, which is defined in Table 1. Following this partial equilibration, the other half of the water which moves on the first day (i.e. $\alpha/2$), now moves on carrying solute with it. A similar series of steps is used for every successive soil layer. Water moving as bypass flow has the same concentration as the mobile water present in the layer where bypass flow is initiated.

Evaporation

Evaporation from the soil is calculated using relationships derived from Penman (1941). Solute and water movement in response to evaporation is calculated according to the model of Nicholls *et al.* (1982).

Transport to drains

In our experimental work we have assumed that all water draining below 30 cm goes direct to the pipe drainage system without delay. This assumption is possible because down-slope interflow in the plough layer is the major lateral saturated flow process (i.e. a 'perched' water table occurs in this soil) and because experimental microplots have been set up immediately over gravel backfill connecting drains with the plough layer at about 30 cm depth. We have further assumed that the layer depth, L, is 50 mm, so it remains to estimate suitable values for α, β and the soil water parameters θ_e, θ_r, θ_m and θ_{ms}.

Estimation of α

We have considered four methods for estimating the permeability parameter.

1. Soil particle size analysis can be used in conjunction with the correlation proposed by Addiscott & Bland (1988). They suggest the relationship

$$\alpha = 1.0271 - 0.000302\, c^2 \tag{4}$$

   where c = percent clay in the soil.

2. The changes in soil water storage predicted by the model can be compared with measured values and the errors minimized. Thus, the function $S(\theta)$ should be minimized:

$$S(\theta) = \sum_{i=1}^{n} \left[\sum_{j=1}^{z} x_j (\theta_{i,j} - \theta_{1,j})_o - \sum_{j=1}^{z} x_j (\theta_{i,j} - \theta_{1,j})_p \right]^2 \tag{5}$$

   where $\theta_{i,j}$ = water content of layer j at time interval i. The soil is divided into six layers of 50 mm depth; n = number of times for which measurements are made. Subscripts o and p refer to observed and predicted, respectively.

3. Observed daily drainage from plots can be compared with predicted values. The appropriate function to be minimized is

$$E(D) = \sum_{i=1}^{n} [D(i)_o - D(i)_p]^2 \tag{6}$$

   where $D(i)_o$, $D(i)_p$ = observed, predicted plot drainage on day i.

4. Observed cumulative drainage from plots can be compared with predicted values. The appropriate function to be minimized is

$$F(T) = \sum_{i=1}^{n} [T(i)_o - T(i)_p]^2 \tag{7}$$

   where T = cumulative drainage from plot (mm).

In both methods 3 and 4, we used corrected values of measured drainage to remove any systematic error between observed and predicted values. This was done by multiplying values of $T(i)_o$ and $D(i)_o$ by the correction factor

$$f = \sum_{i=1}^{n} [T(i)_p / T(i)_o]. \tag{8}$$

In this way we ensured that only the timing of daily and cumulative drainage relative to rainfall events was compared.

Estimation of β

The method for estimating β is better established (Addiscott *et al.*, 1983; Rao *et al.*, 1982) and can be determined, when certain simplifying assumptions are made. Following Rao *et al.* (1982) the fractional equilibration of solute between immobile and mobile water for a spherical aggregate bathed in a mixed solution of limited volume is given by

$$C_r(t) = C_\infty + C_r(0) - C_\infty \sum_{n=1}^{\infty} \frac{6b(b+1)\exp(-D_r q_r t/A^2)}{9 + 9b + q^2 b^2} \quad (9)$$

where $C_r(t)$ = solute concentration in immobile water at time t

C_∞ = equilibrium solute concentration.

$b = \theta_m/\theta_r$,

D_r = effective diffusion coefficient in aggregate (m^2s^{-1}),

A = aggregate radius (m),

t = time (s),

q_n = positive, non-zero roots of the equation

$$\tan q_n = [3q_n/(3 + bq_n^2)] \quad (10)$$

Then solute concentration in the mobile water, $C_m(t)$, is given by

$$C_m(t) = [\varphi/(1-\varphi)][C_r(0) - C_r(t)] \quad (11)$$

where

$$\varphi = \theta_m/(\theta_m + \theta_r). \quad (12)$$

This analysis would give a value for β, the 'hold-back' coefficient, if aggregates were all of one size, taking $t = 1$ d

$$\beta = 1 - C_r(1)/C_\infty. \quad (13)$$

For a soil which consists of an assemblage of aggregates, a weighted average value must be obtained over the range of aggregate sizes present in the soil.

MATERIALS AND METHODS

Field plot isolation and instrumentation

Detailed description of the larger experiment of which this work is part is to be published elsewhere, but Fig. 2 gives a diagram showing the layout of plots 5 to 8 from this experiment. Each plot is approximately 300 m^2 and hydrologically isolated. Deep seepage beneath the depth of drainage is likely to be small. Each plot is drained by three shallow (45 cm deep) pipe drains connected by gravel backfill to the plough layer, and by a 1 m deep downslope collection drain as shown in Fig. 2.

The soil is clay loam of the Winton series with imperfectly drained gleyed B and C horizons. The site is on an approximately 5% slope. Soil details are given in Table 2.

From each plot a single pipe leads into an instrumentation pit, where plot drainflow is measured using a tipping-bucket flow meter (Field Drainage Experimental Unit, Anstey Hall, Cambridge). A reed switch on the bucket sends an impulse for each tip to a Cristie CD6 data logger (Cristie Electronics Ltd, Stonehouse, Gloucestershire) which records number of tips on an hourly cycle. Rainfall is measured on the same hourly cycle using two tipping-bucket rain gauges. Flow-weighted concentration of any solute leaving the plots was measured with a simple flow-dividing device which

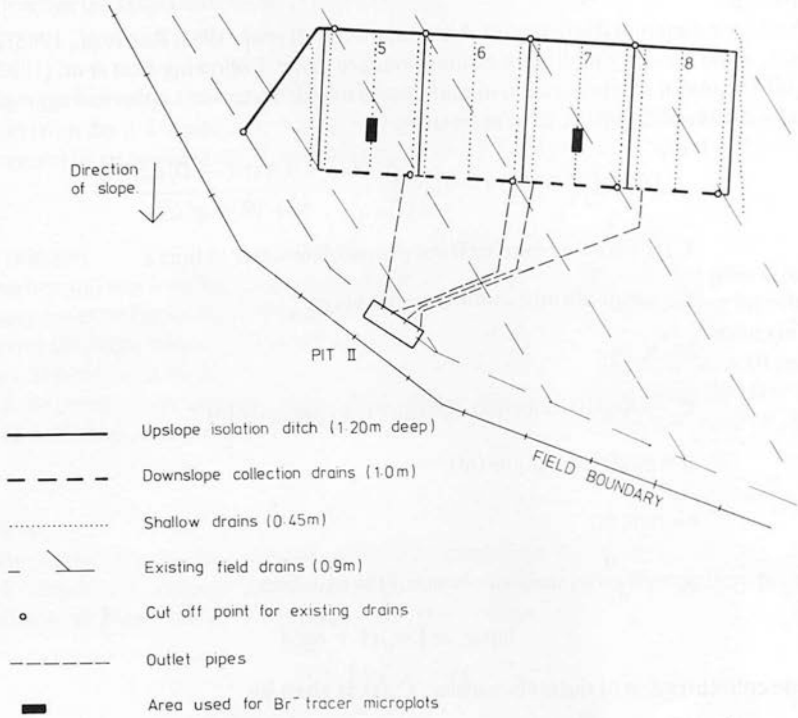

Fig. 2. Hydrological isolation of experimental field plots and siting of microplots at East Flotterstone field.

Table 2. Soil details (plough layer only)

Soil Series	Winton
Soil Association	Rowanhill
Textural class	clay loam
Organic Matter	4.8–6.0%
pH	6.2–6.7
Clay	27%
Classification (USDA)	Typic Haplaquept

takes small samples of the water dispensed as the tipping bucket tips. The samples accumulate so that a single bulk sample for each plot is obtained over each experimental sampling period.

During the experiments some problems were experienced with multiple counting by the reed switch on plot 5. Therefore a correction factor of 0.545, estimated from tipping at low flows, had to be applied to all drainage data from plot 5.

Bromide tracer experiments

The leaching model was evaluated by establishing two 1 m^2 microplots directly over the shallow drains in the centre of plots 5 and 7, as shown in Fig. 2. Aqueous $CaBr_2$ solution (500 cm^3) was applied to each microplot using a spray gun. Microplots were surrounded with an aluminium frame to prevent runoff or runon to the microplots. The fate of the Br^- tracer was followed by determining loading to the plot drainage system and by soil sampling. Two pairs of runs were done, moving the microplot location along the line of the pipe drain for the second run. The Br^- concentration for the second run was approximately four-fold higher, to minimize any error from residual tracer and to improve accuracy of measurement of Br^- concentration. Details of the experimental runs are set out in Table 3.

The Br^- in the collected drainage water was analysed using an Orion specific-ion electrode. Concentrations were greatly diluted by the water draining from the rest of the field plot, but this did not affect estimation of leaching, except that some of the concentrations fell below 1 mg dm^{-3} (the value of the minimum standard) when non-linearity of the electrode response curve may occur. Also, in the first run, allowance had to be made for the chloride interference. A small correction factor was applied to the Br^- concentrations measured. This ranged from 0.8 when [Br] was 0.7 mg dm^{-3} to 0.96 when [Br^-] was 4.8 mg dm^{-3}. To minimize the effects of Cl^- interference, on the second run a larger $CaBr_2$ addition was used (Table 3). Moreover, 500 mg dm^{-3} Cl^- was added to all samples before analysis for Br^- to swamp fluctuations in background Cl^- levels.

The soil in the microplots was sampled to a depth of 35 cm in triplicate at the end of each experimental period using a 6.3 cm diameter intact soil corer. At the end of the second run the soil was also sampled from 35–80 cm, using a Dutch auger, but less than 3% of the total Br^- applied was recovered below 35 cm. The soil core obtained was sectioned into 5-cm sections, dried and sieved before analysis of the [Br^-] in the <2 mm fraction. Stone weights were recorded and [Br^-] was calculated on a mass per unit area basis for each section.

Crop uptake during the experimental period was also measured, and found to be small.

Data for estimating model parameters

The soil moisture characteristic curve for the soil in the plough layer was measured on duplicate samples using standard methods. Changes in soil moisture content were estimated using a Wallingford neutron probe and calibrations were obtained from previous measurements for the same soil series, with appropriate surface correction factors. However, considerable uncertainty about values in the top 10–20 cm existed because of these correction factors. One access tube was located next to each microplot. The soil aggregate-size distribution was determined in duplicate on a sample of field-moist soil which was hand-sieved through a series of mesh sizes from 6 cm to 0.2 cm. (see Table 4).

Actual evaporation from the plots is calculated in the model using input potential evaporation data. The potential evaporation was estimated from a net evaporimeter about 1 km from the experimental site. Monthly average figures were calculated using only days when zero rainfall occurred as discrepancies between site rainfall and rainfall at the evaporimeter were apparent.

MODEL CALIBRATION

Sensitivity analysis

Before considering parameter estimation and comparison between predicted and observed results, the relative importance of perturbations in model parameter values will be considered. Fig. 3 shows a sensitivity analysis for the following default values of the model input parameters, using rainfall and evaporation data from 7 December 1987 to 26 January 1988: $\alpha = 0.2$, $\beta = 0.2$, $\theta_r = 0.287$, $\theta_{ms} = 0.127$, $\theta_e = 0.087$, $\theta_m = 0.042$, $L = 50$ mm, $M = 0.0$ mm.

The percentage of tracer leached below 30 cm is shown as a function of the value of the parameter relative to the default value, with other parameters taking the default value. Note that, since θ_s, the water content at saturation, is given by $\theta_e + \theta_r + \theta_{ms}$, varying one of the water content parameters changes the value of the others unless θ changes as well. The value of θ_r has been varied while keeping the total water content at saturation constant (so θ_{ms} has decreased) but θ_{ms} has been varied while θ_r remains constant. Clearly the model predictions are much more sensitive to α than to any other parameter, except for extreme values. Fluctuations of θ_r and θ_{ms} are likely to be small as these are capacity, not rate parameters, so clearly the major effort in parameter estimation should be in estimation of α.

Estimation of α

Method 1. Addiscott & Bland (1988) suggested the correlation between α and the percent clay in the soil given by Equation (4). Since this soil has about 27% clay the value of α obtained is 0.81.

Table 3. Microplot experimental details

	Run 1: (14/12/87–26/1/88)		Run 2: (26/1/88–29/3/88)	
	Plot 5	Plot 7	Plot 5	Plot 7
Br^- applied ($g\ m^{-2}$)	100.0	100.0	432.2	432.2
Br^- recovered				
soil (%)	10.3	10.0	41.7	18.5
drainage (%)	70.6	77.3	34.3	66.0
total (%)	80.9	87.3	76.0	84.5
Estimated potential evaporation (mm)	13.8	13.8	52.6	52.6
Rainfall (mm)	169.4	169.4	185.9	185.9
Measured field plot drainage (mm)	157.7	145.9	88.0	110.0

Table 4. Aggregate-size distribution

Mesh size (cm)	Cumulative percentage of soil retained
6	38.9
2.7	58.2
0.475	93.5
0.2	98.5
0.1	99.9

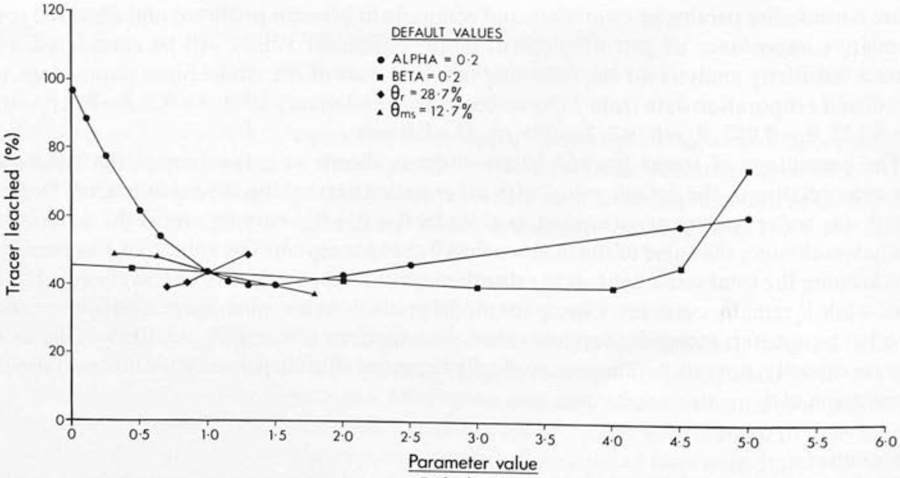

Fig. 3. Sensitivity analysis for leaching model using rainfall data from run 1.

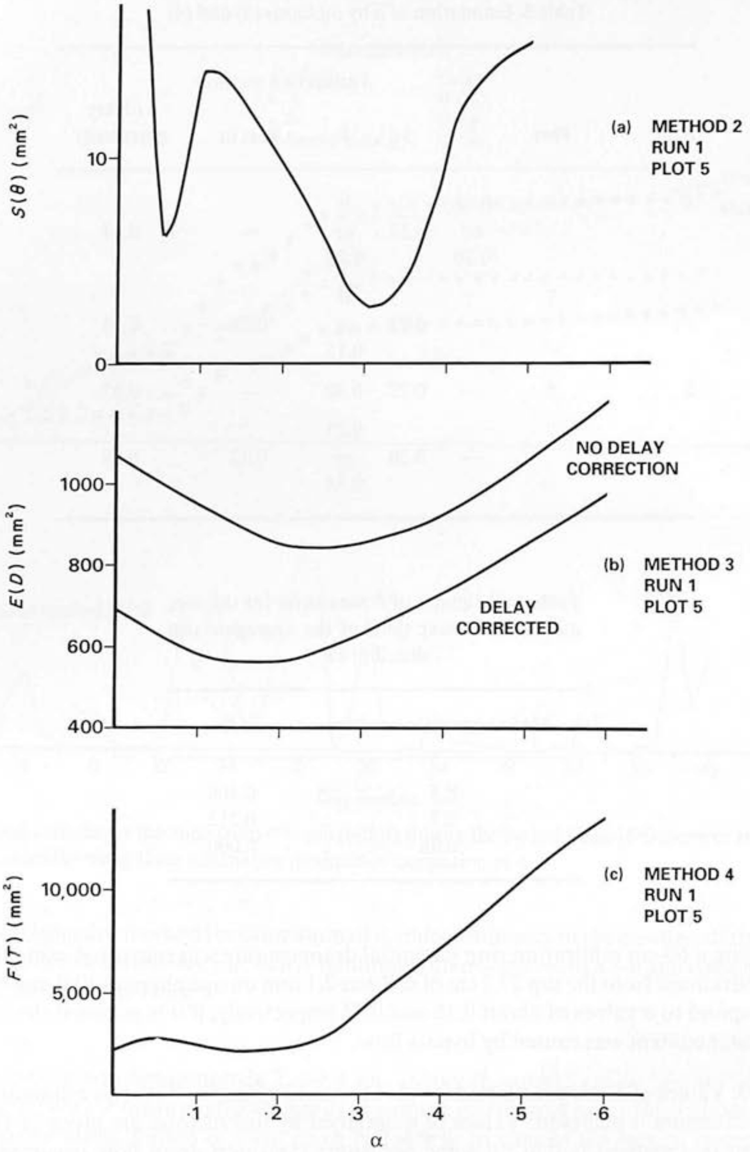

Fig. 4. Estimation of α parameter using least-squares error minimization with three alternative methods. (a) Method (2): soil water content data, see Equation (5). (b) Method (3): daily drain flow data, see Equation (6). Corrected and not corrected for delay in drainage reaching drain outfall. (c) Method (4): cumulative drainflow data, see Equation (7).

Method 2. Values of $S(\theta)$ were calculated using neutron probe data for four dates between 14 December 1987 and 5 February 1989. These are plotted as a function of the assumed value of α in Fig. 4a. Two minima occur and there is no obvious way of deciding on a best estimate of α. When rainfall is frequent, the changes in soil moisture during the winter period are small, and, unless very frequent neutron probe measurements are done, it is clearly difficult to estimate a value for α. Neutron probe data are also unreliable in the surface soil, so the method appears difficult to apply. In February 1989 an alternative approach was investigated, using a gamma probe collimated to allow a resolution of 2 cm. This meant that water contents (assuming bulk density is constant) could be estimated to within 1–2 cm of the soil surface. Following ponding of approximately 50 mm on

Table 5. Estimation of a by methods (3) and (4)

Run	Plot	Estimation method 2	3	4	Best fit	3 (delay corrected)
1	5	0.05 or 0.30	0.27	0 or 0.22	—	0.14
	7	—	0.22	0 or 0.15	0.06	0.14
2	5	—	0.22	0.40	—	0.07
	7	—	0.20	0.05 or 0.68	0.12	0.08

Table 6. Estimates of β parameter for the top, middle and lower third of the aggregate size distribution

Mean aggregate size (cm)	β
8.5	0.466
3.7	0.212
0.96	0.000

two plots using a 60-cm infiltration ring the initial drainage rate was measured using this method. Over 3 d the drainage from the top 27.5 cm of soil was 2.1 mm on one plot and 10.0 mm on the other. These correspond to a values of about 0.16 and 0.76 respectively, if it is assumed that none of this change in water content was caused by bypass flow.

Method 3. Values of $E(D)$ are plotted for plot 5, run 1 in Fig. 4b. With this estimation method a fairly clear minimum is achieved. Values of a obtained by this method are given in Table 5. One problem with this method is that on some occasions measured drain flow occurred in the day following the measured rainfall, particularly if a storm event occurred close to midnight. We have attempted to account for this delay (which occurs because water must travel down the profile and laterally to drains) by the following argument. For run 1, plot 5 the optimized value of a is 0.27 (Table 5). If this represents an upper limit to matrix permeability, then any rainfall at a rate in excess of this value (equivalent to about 1.3 mm d^{-1}) will initiate bypass flow once the storage capacity of the top soil layer has been filled. If bypass flow occurs and if vertical movement of water in the topsoil is rapid, then any transformation of the hydrograph observed in the plot hydrograph should be the result of the travel-time distribution from the base of the plough layer to the drain. By observing rainfall events in which intensity exceeds 1.3 mm d^{-1} a probability density function of travel-time from the base of the plough layer to the drains should be obtainable. The median travel-time observed for two such storms was approximately 4 h. To allow for this delay when estimating a, the measured daily drainflow data have been compared with modelled drainflow using daily rainfall calculated on a cycle 4 h earlier. When this procedure is adopted, estimated a values are somewhat lower (e.g. see delayed corrected curve, Fig. 4b).

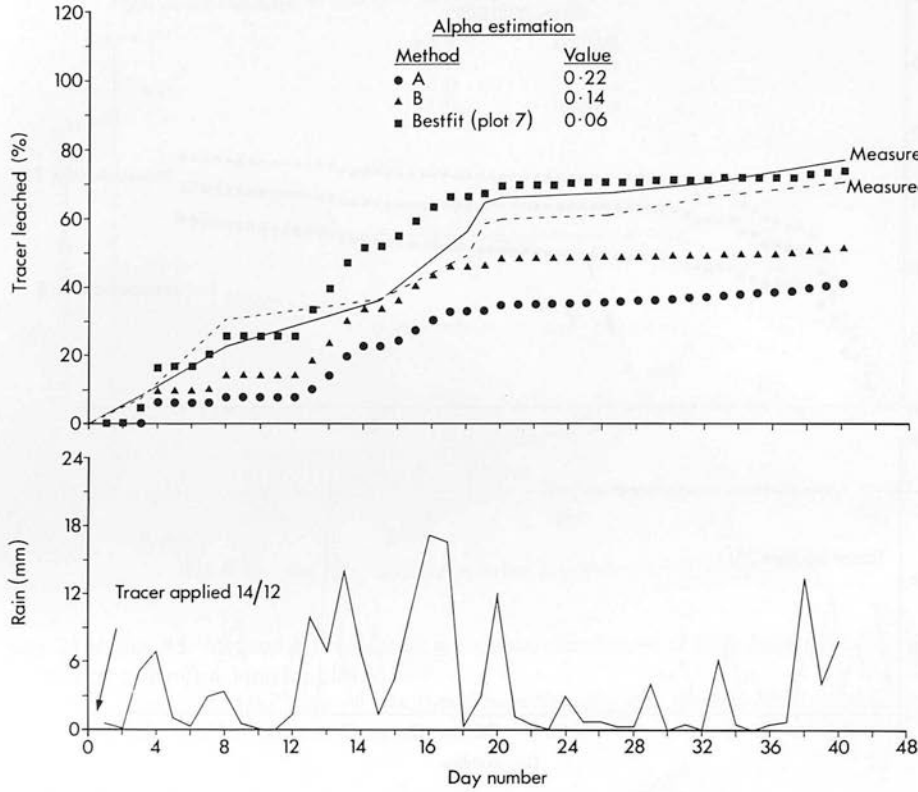

Fig. 5. Measured cumulative leaching from two microplots during the period from 14 December to 25 January, and predicted leaching using three alternative methods of estimation of α.

Method 4. Using this method for estimation of α, indeterminancy in the position of the minimum value of $F(T)$ often occurred (see Fig. 4c). A minimum often occurred at $\alpha = 0$ and elsewhere. Values of α obtained are given in Table 5.

Estimation of β

Using the aggregate-size distribution in Table 4, and taking $D_A = 0.012\ cm^2\ h^{-1}$ for Br^-, the value of β was obtained from the mean aggregate size of the upper, middle and lower third of the cumulative probability distribution. Values of β are given in Table 6. In view of the lack of sensitivity of the model to β a simple mean of these three values was used to describe β for the field soil, giving $\beta \simeq 0.2$ as the default value.

RESULTS AND DISCUSSION

Comparison with experimental data

Figs 5 and 6 shows results of leaching experiments over the two experimental periods. During run 1 (Fig. 5) the weather was almost continually wet and 70–80% of tracer leached. Bromide recovery was good (Table 3) and results for the two plots were in close agreement. Using method 3 to calculate α there was serious underestimation of total leaching loss. The fit was better when the delay in water leaching was allowed for in parameter estimation, but the final leaching prediction is still about 20% low. The best-fit value of α is 0.06 (for plot 7) which is considerably smaller than α estimated by any of the independent methods.

In run 2 (Fig. 6) there was considerably less leaching, with much larger differences between the two plots. In both plots most of the leaching occurred before the extended relatively dry period from

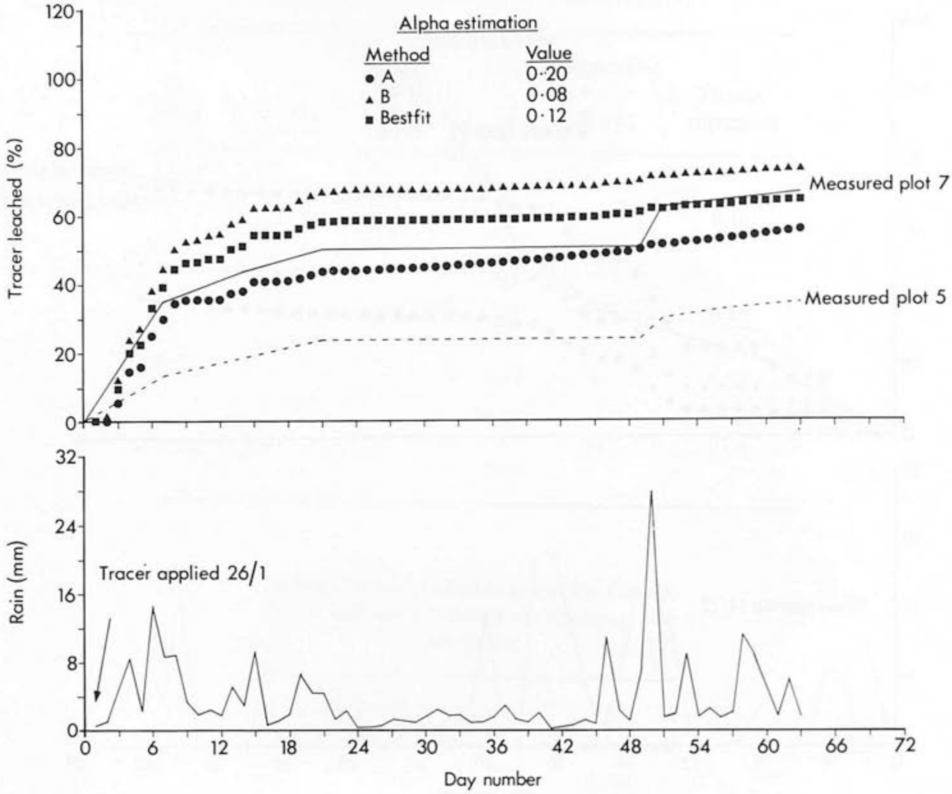

Fig. 6. Measured cumulative Br^- leaching from two microplots during the period from 26 January to 29 March 1988, and predicted leaching using three alternative methods of estimation of a.

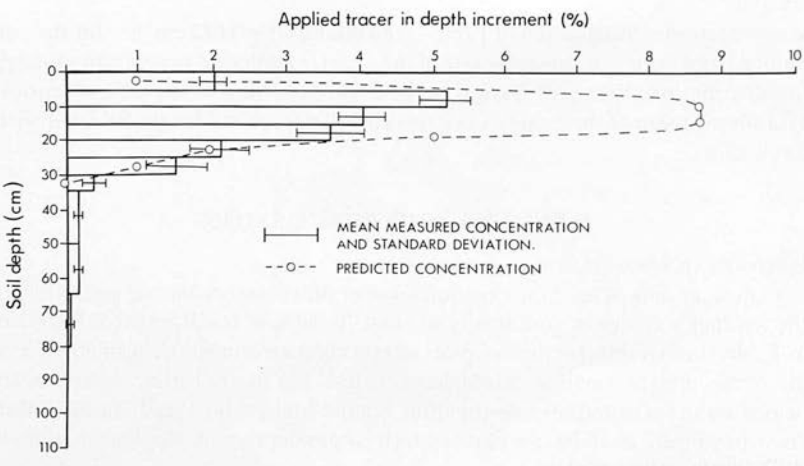

Fig. 7. Mean measured concentration of Br^- in the soil profile after run 2, plot 7, compared with predicted concentration, using $a = 0.08$. Mean values have standard error bars attached.

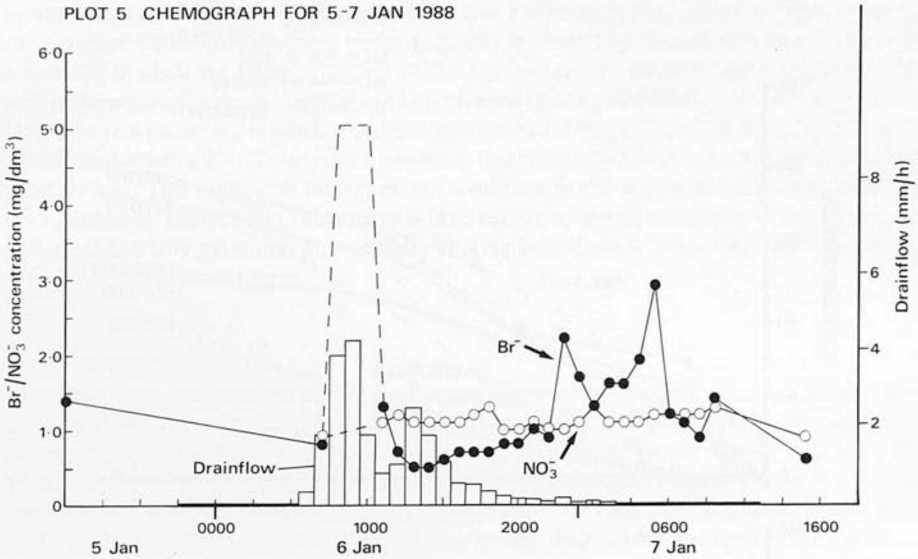

Fig. 8. Br^- and NO_3^- chemographs for a storm event 5–7 January 1988.

day 24 to day 45. Method 3 predictions gave good simulation of data from plot 7, but serious overestimation of leaching for plot 5.

The experimental data show clearly that, when soils are wet, the timing of rainfall relative to fertilizer application is likely to be important.

Model evaluation

Figs 5 and 6 both show that the appearance of solute in the drainage water occurred soon after tracer application. This can only be the result of bypass flow and it is one of the main objectives of this model to predict when such bypass flow occurs in heavy, structured soils. To predict accurately such bypass flow in the present case, quite low values of a are needed, compared with those estimated by Addiscott & Bland (1988). If a is very small, matrix transport of water and solute is very slow. This means that once the top layer of the soil has been leached of solute, very little further predicted leaching occurs. Thus, in Fig. 6, after the extended dry period, the model predicts little further leaching because nearly all the solute has been removed from the top 5 cm layer. The measured leaching from day 48 to 63 was 15% of the total Br^- applied to plot 7. It seems that some mechanism exists whereby solute present in lower layers can reach the drains rapidly. This may be partly because of the imprecise description of equilibration between mobile and immobile water. However, even quite large changes in β have only a small influence on total leaching (Fig. 3). It may be also that rapid bypass flow can pick up solute from lower layers while it is in transit, which is not allowed for in the model. The solute distribution in the soil profile shows that the picture of bypass flow removing solute from surface layers is reasonable since solute concentrations are low both at the top and the base of the plough layer, while a marked peak occurs in the 5–15 cm depth range (see Fig. 7).

Qualitative evidence for the correctness of the bypass-flow model of solute transport and for the assumption that most lateral transport is by plough layer interflow, is provided by chemograph data (Fig. 8). The highest concentration of surface-applied Br^- tracer leaching from the microplot corresponded with the peak on the drain-flow hydrograph*. A small dilution occurred during the second peak, but this was slight. As the two peaks were separated by only 4 h, it seems reasonable that re-equilibration of solute in bypasssing water with mobile water moving through the soil

*The solute concentration was not actually measured during the first hydrograph peak (region of dotted line) but the average concentration during the peak flow period could readily be estimated from the average solute concentration in the flow-weighted water sample for the period 5–6 January. For Br^-, this was 4.0 mg dm^{-3}.

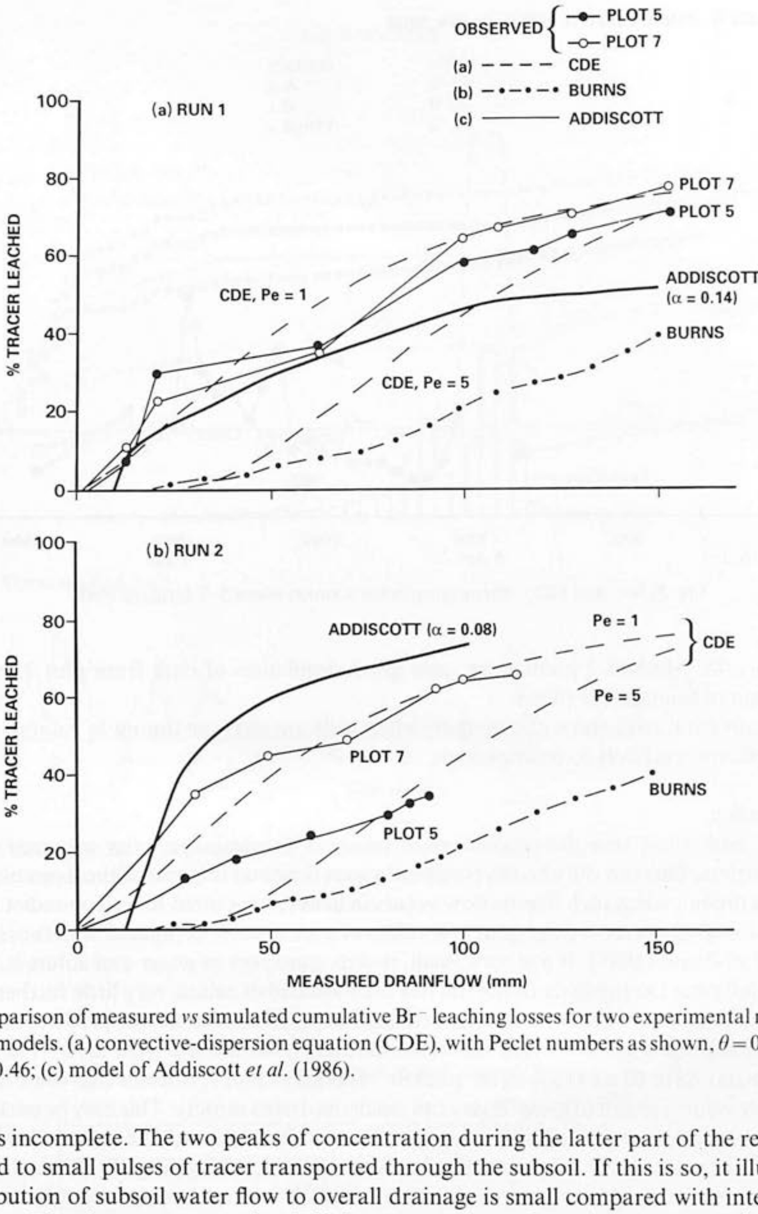

Fig. 9. Comparison of measured *vs* simulated cumulative Br^- leaching losses for two experimental runs and three alternative models. (a) convective-dispersion equation (CDE), with Peclet numbers as shown, $\theta = 0.46$; (b) Burns model, $\theta = 0.46$; (c) model of Addiscott *et al.* (1986).

matrix was incomplete. The two peaks of concentration during the latter part of the recession may correspond to small pulses of tracer transported through the subsoil. If this is so, it illustrates that the contribution of subsoil water flow to overall drainage is small compared with interflow in the plough layer. The nitrate concentration (which represents losses from the whole 300 m^2 plot, and not just the 1 m^2 microplot) was almost unaffected by the change in flow rate. Similar results were obtained from plot 7 during the same storm, except that a peak in $[Br^-]$ occurred during the second peak in the drain-flow hydrograph as well.

The model does not allow for surface runoff at all and clearly this is a major limitation since the route water takes to drainage systems during high intensity rainfall strongly influences the amount of leaching which occurs.

Parameter estimation

The estimation of α is critical to obtaining realistic predictions of solute leaching with the model. However, none of the four methods used is entirely satisfactory in its present state though method 3, using daily drainage data with a 4 h delay, is the best approach for three of the four breakthrough curves measured. The use of the hydrograph to estimate α is attractive because an integrated value can be obtained over a plot or an entire field. In many situations (e.g. where artifical drainage to control 'perched' water tables occurs) such hydrographs should be readily obtainable. However, a more accurate method for transformation of the hydrograph to estimate the travel-time density function to drains may be useful, and this will involve use of a shorter time-interval in the model than 1 d. The value of a estimated is much smaller than expected from Addiscott & Bland's observations on soils of similar texture in the south of England; this probably reflects the weaker structure that exists in this soil. This highlights an important weakness in the approach of Addiscott & Bland (1988) to parameter estimation in that there is little reason to expect texture to be a good predictor for what are essentially structural phenomena. This is particularly so when considering leaching of recently applied fertilizer nitrogen.

Criteria for success

As this is a management model we should apply a management criterion for success. The objective of this modelling exercise, as stated in the Introduction, was to improve on present methods of predicting the proportion of fertilizer N leached from early spring applications to winter or spring cereals. This could then be used to make adjustments to further applications if necessary. Current fertilizer recommendations in Scotland suggest that, if rainfall exceeds 150 mm on loamy soils, leaching losses of early N will be 10% for winter cereals or 15% for spring cereals. If we assume that all the N lost is from the nitrate-N applied as fertilizer then this corresponds to 20–30% of the total nitrate-N application, if NH_4NO_3 is used. Measured losses of the Br^- tracer varied from 34–77% for the two runs (Table 3), which suggests that current recommendations seriously underpredict leaching losses. Model predictions, using method 3 with delayed drainage to estimate a, give 52% loss for run 1 and 70% for run 2. Therefore, though the leaching losses in run 2 are variable, the model gives a better prediction than current recommendations. These also take no account of timing of rainfall, which model predictions show to have a strong influence on losses incurred, espcially when a is small.

Comparison with other models

There are many models of varying complexity available to simulate solute transport processes in soils (see Addiscott & Wagenet, 1985). Two other models that have been applied to approximate field-scale simulation (Cameron & Wild, 1982), are the chromatographic plate model of Burns (1974) and the convective-dispersion equation (Rose *et al.*, 1982) applied assuming constant water content in the transport zone and steady-state flow conditions. The predictions of leaching using these two models have been compared with measured data, and with predictions using the present model. a was estimated using method 3 with drainage delay correction. For the convective-dispersion equation, type A-1 boundary conditions have been used (van Genuchten & Wierenga, 1986). Comparison of predictions with measured leaching *vs* measured drainflow (Fig. 9) show poor predictions by the Burns model for both runs. The convective-dispersion equation gives rather good simulation if a suitable value of the Peclet number is chosen. Peclet numbers of 1 and 5 have been used (leaching is assumed to occur over 30 cm, so these correspond to dispersion lengths of 30 cm and 6 cm respectively). This suggests that estimation of a characteristic dispersion length and application of a steady-state convective-dispersion leaching model may be as effective a way of modelling the leaching from the top 30 cm of soil as using the model of Addiscott *et al.* (1986), given the difficulties in obtaining suitable independent estimates of a.

Further work

The calibration and validation work reported in this paper refers only to vertical transport in the plough layer. In drained soils on heavy land there is an important component of lateral flow that has not been considered. This is at present under study using whole-plot tracer applications. Poor resolution of the neutron probe in the surface soil is a possible explanation for the failure of method 2 for estimation of a and work is currently in progress to investigate the use of a γ-densitometry probe to measure water content changes in the surface layers of soil during internal drainage.

CONCLUSION

The model of Addiscott *et al.* (1986) for approximate estimation of leaching in heavy, structured soils has been assessed using a series of field leaching experiments. The basic physical description on which the model is based can predict the general form of the solute breakthrough and the soil profile solute distribution in these experiments. Absolute prediction of leaching was accurate to within $\pm 20\%$ in three cases, but in one case were much less accurate. However, the method is an improvement on the existing method. As a management model, it may be sufficiently accurate to predict leaching of fertilizer NO_3^- after application. The estimation of the hydraulic conductivity parameter, a, is critical to model predictions and no fully satisfactory method has been found to estimate it. However, use of the daily drainage hydrograph from field plots to estimate a seems a promising approach, but it may be necessary to use hourly rather than daily hydrological data, which will mean modification of the model. Experimental data show the vulnerability of recently applied fertilizer to leaching, especially when the soil is wet.

ACKNOWLEDGEMENTS

The use of the model SLIM, supplied by Tom Addiscott, is gratefully acknowledged. We would also like to thank Shirley Hannan for preparing the manuscript for publication.

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(*Received 11 April 1989; accepted 11 January 1990*)

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REDMAN, M.H. Ph.D. 1992

Measurement of nitrate leaching losses from arable plots under different nitrogen input regimes

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Abstract. Leaching losses of nitrate-nitrogen were measured from a set of eight hydrologically isolated plots on a clay loam soil over the period from September 1987 to February 1990. Variable drainflow recovery from the plots hampered accurate estimation of nitrate loading, but results suggest that, when inorganic nitrogen fertilizer is applied up to the recommended amount, there is little influence of the amount applied on the amount leached. We did, however, observe the following effects on nitrate leaching: leguminous green manure incorporated in autumn increased leaching of nitrate–nitrogen by 10–15 kg per hectare during the winter; autumn cultivation caused some increase in leaching compared with no cultivation in one year; some systematic variations in nitrate leaching occurred between years and between plots, but were unrelated to treatments.

From the results we conclude that green manuring does not provide sufficient nitrogen for organically grown crops on this soil but contributes significantly to nitrate leaching, and that growing spring cereals, with the land remaining in stubble as long as possible in autumn, may be the best strategy to minimize nitrate leaching.

INTRODUCTION

RECOVERY of fertilizer nitrogen, determined by ^{15}N tracer experiments on heavy Scottish soils, is usually only about 40–50% of application rates (Smith *et al.*, 1984), and may be as little as 15% in unfavourable summers (Smith *et al.*, 1988). The extent to which poor recovery is due to pool substitution of fertilizer and biomass nitrogen (Jenkinson *et al.*, 1985), to denitrification or to leaching is still a matter of uncertainty. Other workers are seeking to quantify biomass nitrogen turnover (Rees, 1989) and denitrification on these soils; this paper deals with nitrate leaching. Pressure to improve fertilizer use efficiency and decrease nitrate leaching losses from agricultural land (Department of the Environment, 1986) means that we also need to assess the impact of alternative fertilizer practices on decreasing leaching losses. For these reasons we established in 1987 a set of eight hydrologically isolated plots on a clay loam soil, so that leaching losses from arable plots receiving different fertilizer treatments (both inorganic and organic forms of nitrogen) could be measured directly to quantify losses of fertilizer and soil nitrogen.

Nitrate leaching from arable land has been measured over at least ten years at Brimstone Farm, Oxon, as part of a cultivation and drainage experiment (Cannell *et al.*, 1984; Harris *et al.*, 1984; Dowdell *et al.*, 1987). Long-term measurements of soil water quality have also been made using lysimeters at Rothamsted (Lawes *et al.*, 1881, quoted in Addiscott, 1988) and at Aberdeen (Hendrick, 1930) (see Table 5). The Aberdeen results showed very much smaller long-term annual leaching losses than at Rothamsted, suggesting that losses of mineralized N from arable land could be less important in the cooler climate typical of northern Britain than in the south. The seasonal pattern of loss may also be very different, as fertilizer is often applied in Scotland to soils that are still at or above field capacity and susceptible to leaching. However the Aberdeen data is up to 70 years old. Therefore it is important to obtain good up-to-date regional information about the effect of fertilizer practice on nitrate leaching.

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MATERIALS AND METHODS

Site description

The site chosen was at Glencorse Mains Farm, near Penicuik, Midlothian (National grid reference NT236627). The altitude is 207 m O.D. The land capability class for Agriculture is 3.2 (Bibby *et al.*, 1982) and the field had previously been under winter barley. A new drainage scheme had been installed in September 1986 and plans were available. The soil is an imperfectly drained clay loam of the Winton Series. Samples were taken for routine physical and chemical analysis in March 1987. Details of the profile are given in Table 1. Subsoil texture is quite variable (sandy loam to silty clay loam), as in many soils derived from glacial deposits.

Plot isolation

In artificially drained arable fields, the best possible estimate of nitrate leaching to surface water systems is obtained by hydrological isolation of a known area of land. Without hydrological isolation, the area contributing to leaching is unknown and water and solute mass balance will be difficult to obtain. Having established the suitability of the soil for

Table 1. Typical soil profile description

Horizon	
Ap(g)	0–30 cm: Dark greyish brown (10 YR 4/2) with few medium yellowish brown (10 YR 5/6) mottles; clay loam; moderate medium subangular blocky; firm; moist; low organic matter; common fine roots; common small subangular stones; clear change to:
B(g)	30–72 cm: Brown (7.5 YR 5/2) with common medium and fine yellowish red (5 YR 5/8) mottles and few fine grey (10 YR 6/1) gley streaks; silty clay loam to clay loam; weak medium angular blocky; firm; moist; no organic matter; very few fine roots; few small subangular stones; gradual change to:
B_2(g)	72–82 cm: Dark brown-brown (7.5 YR 4/4) with common fine strong brown (7.5 YR 5/8) mottles and common to many medium and coarse grey (N/6) gley patches and streaks; silty clay loam; weak medium angular blocky to massive; firm; moist; no organic matter; no roots; few small subangular stones; clear change to:
BC(g)	82–110+ cm: Reddish brown (5 YR 4/4) with common fine strong brown (7.5 YR 5/8) mottles and many coarse grey (N/6) gley patches; common small blocky manganiferous concretions; sandy clay loam; weak medium subangular blocky to massive; firm; moist, but becoming wet; no organic matter; no roots; common medium stones; few large stones; mainly sandstone with some igneous.

Soil series: Winton
Soil Association: *Rowanhill*
USDA Classification: *Typic Haplaquent*

Horizon	pH	Organic matter (%)	Mechanical analysis range (eight samples)			Bulk density at time of sampling g/cm^3
			Sand (%)	Silt (%)	Clay (%)	
Ap(g)	6.2–6.7	4.8–6.0	31–53	25–44	21–25	1.19
B(g)	6.3–6.7	0.6–1.9	14–57	21–52	17–33	1.47

hydrological isolation (i.e. that deep percolation of drainage water beyond the depth of isolation would be slow), work began in March 1987 to install a set of hydrologically isolated plots. This was done as follows (see also Figs 1 and 2).

(1) Two isolation ditches, nominally 1.2 m deep, were excavated to prevent inflow of water to the plot area from upslope. These cut the existing drainage system and a number of old clay tile drains which ranged in depth from 0.5 to 1.0 m. A 100-mm pipe was laid in the trenches, which were polythene lined on the downslope side to prevent drainage of water from the plot area. The trenches were then backfilled with gravel to the soil surface. The pipe was led to a suitable outfall in the existing drainage system.

(2) At the bottom edge of each of the eight plots a drain 1.0 m deep was installed, which cut the existing drainage system. The drain was polythene lined on the downslope side. A 40-mm plastic drainage pipe, with gravel backfill to the soil surface, collected drain water. Each plot drain was connected, using 40 mm closed plastic pipe, to an instrument pit where drainflow and water quality could be measured.

(3) Three slit trenches were cut to a nominal depth of 50 cm perpendicular to the plot drains in each plot as shown in Figure 2. One of these ran up each edge of the plots and one ran up the middle of the plots to give a nominal spacing of 7 m. Forty-millimetre pipe was laid in these trenches and connected with the plot drains, and the trenches were backfilled to the soil surface. These shallow drains, separating plots from one another, ran almost exactly perpendicular to the contours. As the soil was imperfectly drained we considered this depth was sufficient to prevent lateral water movement between plots. There was a guard strip of approximately 1 m between plots, to which no fertilizer was applied.

Hydrological measurements

Rainfall was measured using two tipping bucket rain gauges mounted on concrete plinths at either end of the central discard area between the blocks. The drainage from each plot was piped to an instrument pit, in which tipping bucket flow meters (Field Drainage Experimental Unit, Anstey

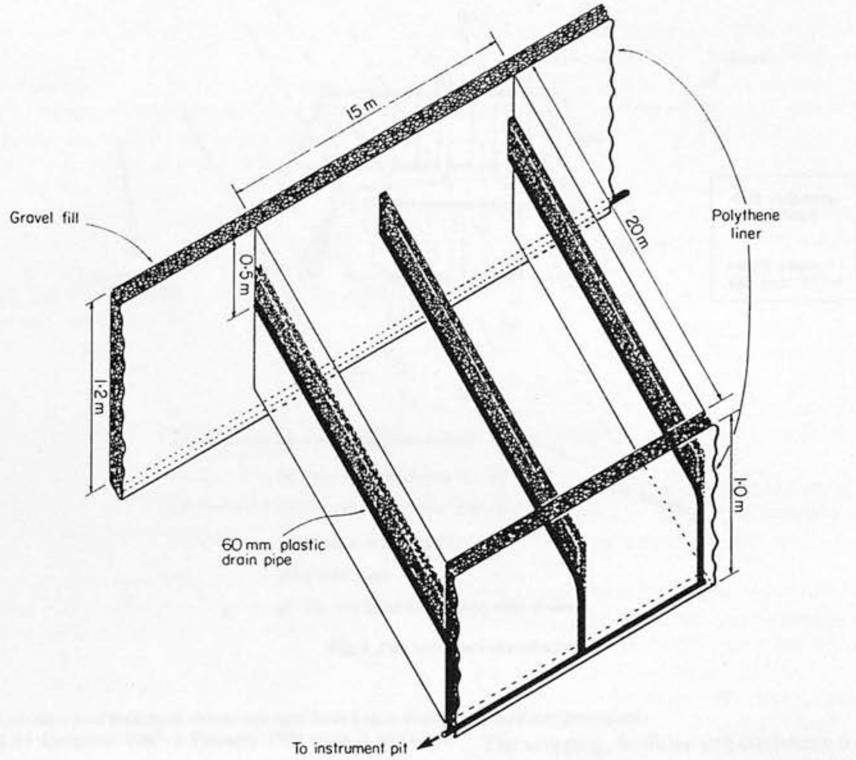

Fig. 1. Plot isolation – single plot.

Hall, Trumpington, Cambridgeshire) were installed, mounted on a concrete base. The tipping was registered by a reed switch connected to one of two Christie CD6 data loggers (Christie Electronics Ltd, Gloucestershire, UK), running on an hourly cycle. These loggers were replaced by electronic integral counters in September 1988. Data were collected approximately once a week. The tipping bucket flow meter for Plot 5 gave multiple counts on some occasions, so drainflow from this plot needed to be corrected. The frequency of the occurrence of multiple counting (two or more tips per hour when zero or one is the norm) was estimated from periods of slow flow, to give a correction.

The soil moisture content was measured during the period January 1988–June 1988 using a neutron probe. Access tubes were installed in Plots 1, 3, 5 and 7. Calibrations were obtained from Dr D.B. Naysmith (personal communication).

Drainage water sampling
Two devices were used to sample the drainflow.

(1) An integral sampler. This consisted of a closed plastic bucket with a plastic pipe running through it which carried water discharged from one side of the tipping bucket. A small hole in the pipe allowed some of the water which tipped into the pipe (about 10 ml) to flow into the bucket. Aliquots of water sampled from tipping buckets in this way accumulated in the collecting bucket, and a single sample of the water in the bucket was removed at the end of each sampling period. The volume of each aliquot may vary somewhat but this is unimportant because the number of aliquots collected is so large ($n = 300$ for 10 mm drainflow). This system had a considerable advantage over the second sampling method because only one sample was needed for analysis to obtain an unbiased estimate of the flow-weighted average solute concentration during the sampling period.

(2) A spot sampler. This was a bottle sampler with electronic timing device (Automatic Liquid Samplers SEC2 model 24) for sampling at fixed time intervals. Each sample was *c.* 0.5 litre and was taken from a receptacle into which the tipping bucket discharged. This was designed so that the sample taken was not severely contaminated by stagnant water remaining from previous tips. This system was used only for detailed chemograph studies reported elsewhere (Vinten & Redman, 1990) and for checking the values of the concentrations obtained using the flow-weighted integral sampler. Table 2 gives estimates of total loading of nitrate–nitrogen from two plots made during the period 14 December

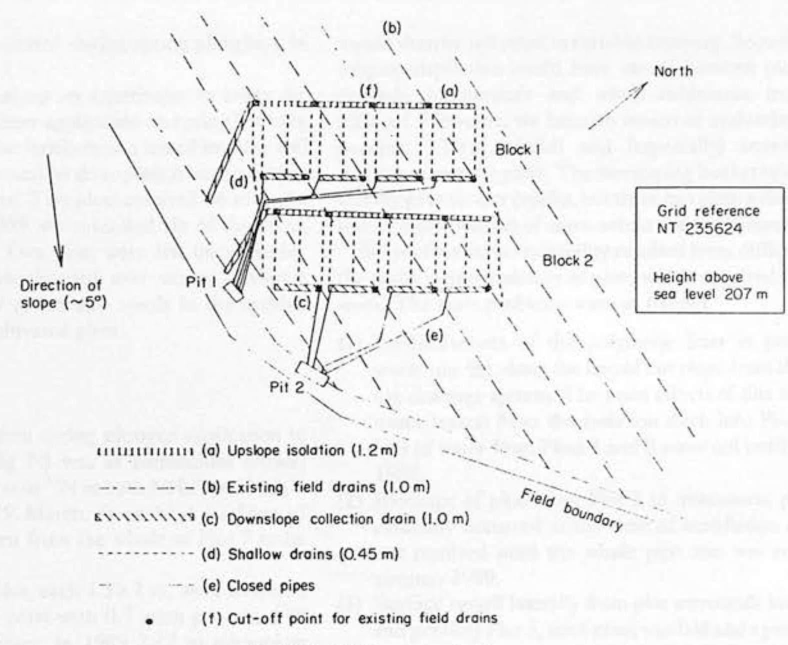

Fig. 2. Plot isolation – overall scheme.

Table 2. Estimated total loading of nitrate–nitrogen from 2 plots during the period 14 December 1987–2 February 1988 using 3 alternative methods

Estimation method	Plot 5 (kg/ha)	Plot 7 (kg/ha)
Simple average of spot samples	2.60	2.23
Flow weighted average using only spot samples	2.29	2.19
Flow weighted average using integral sampler	2.20	2.16

1987–2 February 1988, using three alternative methods. It shows good agreement between flow weighted values using the spot sampler and results using the integral sampler.

Fertilizer treatments

The cropping, fertilizer and cultivation treatments applied to the eight plots are summarized in Table 3. The objective of the first two years was to measure losses from arable land fertilized by broadcasting zero, low or recommended levels of nitrogen fertilizer, and to compare these with losses from plots fertilized by ploughing in a green manure crop (forage peas—*Pisum arverse* var. Birte). This crop was grown in 1987 on Plots 3 and 8. It was cut and incorporated in the autumn of 1987 to provide nitrogen to the following winter barley crop. Full details are given in Redman *et al.* (1989). A cover crop of winter rye (*Secale cereale* var. Rheidol) was established in autumn 1988 on Plots 1 and 6 to take up mineralized nitrogen and so limit winter leach-

Table 3. Cropping and fertilizer treatments 1987–1989

	1987		1988		1989	
	Spring	Autumn	Spring	Autumn	Spring	Autumn
Plot						
1	SB (0)	WB (0)	WB (75B)	RC	SB (120M)	F
2	SB (120B)	WB (0)	WB (0)	F	SB (120B)	CP/SS
3	PEAS (20B)	WB (0)	WB (0)	F	SB (120B)	CP/SS
4	SB (60B)	WB (0)	WB (150B)	F	SB (0)	CP/SS
5	SB (120B)	WB (0)	WB (0)	F	SB (120M)	CP/SS
6	SB (0)	WB (0)	WB (75B)	RC	SB (120B)	CP/SS
7	SB (60B)	WB (0)	WB (150B)	F	SB (0)	CP/SS
8	PEAS (20B)	WB (0)	WB (0)	F	SB (120M)	F

SB = Spring barley; WB = Winter barley; F = Fallow (Uncultivated); RC = Winter Rye Cover Crop; 120 = represents N fertilizer application kg/ha; M = fertilizer mixed into soil; B = fertilizer broadcast; CP/SS = plot chisel ploughed and subsoiled in autumn.

ng losses. It was incorporated during spring ploughing in 1989.

In spring 1989 we set up an experiment to study the effect of method of fertilizer application on spring leaching losses. On three plots the fertilizer was mixed into the soil at the recommended rate and on three plots it was broadcast at the recommended rate. Two plots received no nitrogen fertilizer. In autumn 1989 we subsoiled six of the plots, after chisel ploughing. Two plots were left uncultivated. Plots left fallow and uncultivated over winter carried a considerable amount of clover and weeds in the stubble compared to autumn cultivated plots.

^{15}N tracer experiments

On plot 7 in 1988 the first spring nitrogen application to the winter barley (75 kg N) was as ammonium nitrate, containing 0.7 atom per cent ^{15}N in both NH_4^+ and NO_3^-. This was applied on 29 March. It enabled leaching of fertilizer-derived nitrogen from the whole of Plot 7 to be measured directly.

Two microplots per plot, each 1.5×2 m, were fertilized in each of the first two years with 0.7 atom per cent ^{15}N ammonium nitrate fertilizer. In 1989 2×2 m microplots were used where fertilizer was mixed in. Methods for herbage analysis are given in Smith *et al.* (1984).

Water sample analysis

Nitrate and ammonium concentrations in the water samples were determined by continuous flow analysis using the methods of Hendrickson & Selmer Olsen (1970) and Crooke & Simpson (1971), respectively. NH_4^+ amounts were usually insignificant.

The ^{15}N abundance in samples of leachate from Plot 7 was determined by mass spectroscopy of acidified samples concentrated on a sand bath. Distillation with magnesium oxide gave the NH_4^+ fraction and subsequent redistillation with Devarda's alloy gave the NO_3^- fraction.

RESULTS AND DISCUSSION

Water balance

Drainflow recovery from the eight plots was initially very variable (Table 4). This variability derived from several sources. First, subsoil hydraulic conductivity is known to be quite variable. Snaebjornsson (1977) found that saturated conductivity (K_{sat}) of the 50–75 cm depth range of this soil series has a median value of 3 mm per day with a 67% range of 1.3–7.6 mm per day. Analysis of the intensity of rainfall during the period November 1987–March 1988 showed that 45% of the daily rainfall had an intensity less than 7.6 mm per day but only 5% had an intensity less than 1.3 mm per day. Thus, a significant but quite variable proportion of the winter rainfall could be lost by deep percolation without the development of a perched water table draining to the artificial drainage system. Variability of K_{sat} would then be reflected in variable recovery. Second, actual evapotranspiration could have varied between plots, particularly in summer and when cultivation treatments differed. However, we have no means of evaluating this at present. Third, rainfall and (especially) snowfall was variable across the plots. The two tipping bucket rain gauges usually gave similar results, but there was often a clearly non-uniform distribution of snow across the experimental area.

Most of the initial variability resulted from difficulty with the hydrological isolation of plots which required remedial work. The main problems were as follows.

(1) Ineffectiveness of the polythene liner in preventing water moving along the line of cut pipes from the existing drainage system. The main effects of this were that water leaked from the isolation ditch into Plot 6, and loss of water from Plots 4 and 8 occurred until autumn 1988.
(2) Blockage of pipe from Plot 3 to instrument pit. This evidently occurred at the time of installation and was not resolved until the whole pipe run was re-laid in autumn 1989.
(3) Surface runoff laterally from plot surrounds into Plot 1 and possibly Plot 5, until grass was laid and a permanent surface interception system was installed along the edges of these plots in autumn 1988.
(4) Leakage of water into plots from diagonal drains passing close to the southern corners of Plots 1 and 5 (see Fig. 2). These drains were rerouted in August 1990.

There was also less recovery in Block 1. This suggests that there is a systematic decrease in subsoil K_{sat} from the top to the bottom of the experimental area.

Soil hydrological measurements

Although the approach used in this work is primarily a 'black-box' approach to measuring leaching of nitrate, some measurements of soil hydrological processes were made. These processes have also been discussed in other work (Vinten *et al.*, 1991; Vinten & Redman, 1990). We report some data here to illustrate the behaviour of water in this soil. Figure 3 shows a typical hydrograph from Plot 7. Most incident rainfall was recovered, and there is a sharp recession after rainfall ceases. Most of the drainage occurs with a small recession constant (m = 1.8 h), but the last 1–1.5 mm of drainage occurs with a slower recession (m = 6.7 h). This probably reflects a change in the major pathway of water from topsoil to subsoil, though the later part of the recession may also be influenced by slow internal drainage of the topsoil following wetting (Vinten & Redman, 1990). The specific yield of the subsoil was estimated from combined water table recession and drainflow recession data to be about 0.016 cm^3 per cm^3.

Figure 4 gives the results of neutron probe measurements (4 replicate tubes) over the period from 20 January 1988 to 24 June 1988. Little drying out of the soil by winter barley occurs before mid-April, and the depth of removal

Table 4. Drainflow recovery, nitrate concentrations and estimated nitrate leached during five six-month periods 1987–90. The uncorrected losses do not include corrections for guard area or drainflow recovery; the corrected losses were calculated using Equation 1

	Plot†	N fertilizer (kg/ha)	Crop/ cultivation	D (mm)	$[NO_3]$ (mg/l)	Nitrate leached (kg/ha) (Uncorrected)	(Corrected)
Winter 87/88	1	0	Winter Barley	594	2.2	12.8	5.8
	6	0	Winter Barley	679	4.0	27.2	11.2
R = 507 mm	4	60	Winter Barley	95	4.5	4.3	15.7
D = 346 mm	7	60	Winter Barley	346	2.1	7.2	6.8
ETp = 162 mm	2	120	Winter Barley	447	3.0	13.2	8.7
	5	120	Winter Barley	421	2.5	10.7	6.7
	3	20(P)*	Winter Barley	49	9.4	4.6	26.7
	8	20(P)	Winter Barley	136	6.4	8.7	22.7
Summer 88	2	0	Winter Barley	212	6.1	13.0	10.0
	5	0	Winter Barley	169	4.8	8.1	6.0
R = 499 mm	1	75	Winter Barley	305	7.1	21.7	16.6
D = 195 mm	6	75	Winter Barley	353	3.5	12.4	6.4
ETp = 401 mm	4	150	Winter Barley	217	6.2	13.4	11.8
	7	150	Winter Barley	193	4.7	9.0	8.4
	3	0(P)*	Winter Barley	9	5.6	0.5	6.4
	8	0(P)	Winter Barley	100	1.9	1.9	3.3
Winter 88/89	2	0	Stubble	259	2.5	6.5	8.0
	5	0	Stubble	712	1.4	10.1	6.0
R = 481 mm	1	75	Rye	406	0.9	3.6	2.7
D = 359 mm	6	75	Rye	306	1.1	3.3	3.4
ETp = 180 mm	4	150	Stubble	342	4.9	16.7	16.6
	7	150	Stubble	353	1.7	6.1	5.7
	3	0	Stubble	5	2.4	0.1	5.0
	8	0	Stubble	496	1.7	9.8	5.3
Summer 89	4	0	Spring Barley	90	6.3	5.7	6.2
	7	0	Spring Barley	85	4.6	3.9	3.7
R = 390 mm	1	120 M	Spring Barley	105	3.6	3.8	4.0
D = 111 mm	5	120 M	Spring Barley	236**	10.0	23.5	7.5
(excl 5)	8	120 M	Spring Barley	139	7.1	9.8	6.8
ETp Not available	2	120 B	Spring Barley	50	7.8	3.9	7.1
	3	120 B	Spring Barley	—	—	—	—
	6	120 B	Spring Barley	74	9.5	7.0	7.1
Winter 89/90	4	0	CP/SS	383	10.1	38.7	51.1
	7	0	CP/SS	507	5.7	29.0	29.2
R = 560 mm	2	120 B	CP/SS	257	7.7	19.7	36.3
D = 482 mm	3	120 B	CP/SS	341	6.4	21.7	34.1
ETp Not available	5	120 M	CP/SS	—	—	—	—
	6	120 B	CP/SS	476	5.4	25.6	30.1
	1	120 M	Stubble	887	2.5	22.4	16.6
	8	120 M	Stubble	520	4.0	20.6	20.1

†Plot 1: prone to high recoveries. Surface run-off interception improved; autumn 1988—may also have greater snow accumulation than other plots.
Plot 2: functions well throughout, though recovery low in winter 1989/90.
Plot 3: remedial work successful, autumn 1989.
Plot 4: losses from eastern corner stopped autumn 1988.
Plot 5: initially good recoveries, but began to collect extra water after remedial work on Plot 6. Multicounting also occurs.
Plot 6: initially high recoveries, remedial work successful autumn 1988.
Plot 7: used as benchmark throughout experimental period for correcting drainflow (consistently close to mean recovery).
Plot 8: losses from eastern corner stopped autumn 1988.
*(P): previous crop peas.
**: rainfall is mean of two rain gauge values. Drainflow is mean of eight plots. Potential evapotranspiration was estimated by the method of Thornthwaite (1948).

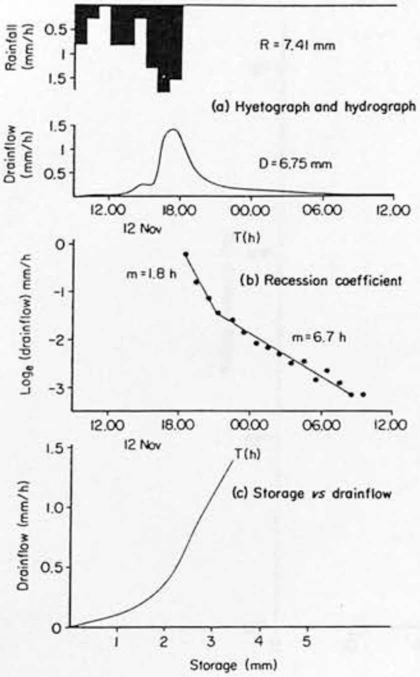

Fig. 3. Drainflow hydrograph data from storm on 12–13 November 1989, Plot 7. (a) Hyetograph and hydrograph. (b) Determination of recession coefficients from log-linear plot of recession curve. (c) Storage *vs.* drainflow relationship.

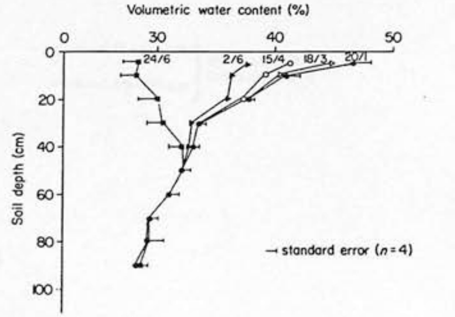

Fig. 4. Volumetric water content profiles on 5 dates: 20 January 1988; 18 March 1988; 15 April 1988; 2 June 1988; 24 June 1988. Means of 4 tubes (Plots 1, 3, 5 and 7).

of water is limited to the top 40–50 cm. The soil moisture deficit developed by 24 June 1988 was 34 mm. Figure 5 gives the moisture release curves for samples of A horizon and B(g) horizon soil. The easily available water capacity (5–200 kPa) of the soil is large (14.6% for topsoil, 13.5% for subsoil), but because rooting depth is often impeded by soil conditions, the soil can be prone to droughtiness. This was especially evident in the summer of 1989, when spring cultivation caused the development of a serious pan and the summer was very dry.

Nitrate concentrations and loading

There was much variability in nitrate concentrations between replicate plots. However, the nitrate concentrations draining from the plots on which the pea green manure was incorporated were greater than from the other treatments (Table 3). To calculate nitrate loading, leaching losses had to be corrected to a standard drainflow. To do this, Plot 7 was used as a benchmark plot as it had consistent total apparent rainfall recoveries over the experimental period. The following procedure was used to calculate the corrected weekly loading for all plots:

$$L_{corr} = \frac{0.01\,(D_{exp}NA_{tot} - G)}{A_{agr}} \qquad (1)$$

where

L_{corr} = nitrate leached from fertilized plot (kg/ha)
D_{exp} = drainflow from Plot 7 (l/m^2)
N = nitrate concentration (mg/l)
A_{tot} = total plot area (including ditches and guard areas) (m^2)
A_{agr} = plot area to which fertilizer was applied (m^2)
G = correction for loading from the unfertilized guard area (mg)

The decreased drainflow in Plot 3 gave fewer water samples and so fewer nitrate concentration values were obtained. Therefore L_{corr} was not calculated on a weekly basis in Plot 3 but instead over whatever time period was needed to include a nitrate concentration value. This correction procedure involved two assumptions.

(1) For plots with little rainfall recovery (notably Plot 3), the flow weighted mean nitrate concentration of the measured/sampled drainflow represented an unbiased estimate of the flow weighted mean nitrate concentration of the 'expected' drainflow. In other words, it was assumed that there was no significant bias in the estimation of the true mean nitrate concentration of plot drainage water by only sampling drainflow generated by relatively large plot discharges following heavy rain. This assumption was tested using data from spot sampled drainflow from Plot 5 on an 8-hour cycle between 14 December 1987 and 2 February 1988. Investigation of the relationship between drainflow and nitrate concentration of the spot samples suggested little bias in the estimation of mean nitrate concentration if only high flow rates were sampled.
(2) For plots with large rainfall recovery, any water moving into a plot contained the same nitrate concentration as water derived from the plot. This was probably a reasonable assumption for any ground or drain water movement into the plots, given the small range of nitrate concentrations from the different treatments during most of the experimental period (with the exception of Plots 3 and 8 in winter 1987/88). However, if surface

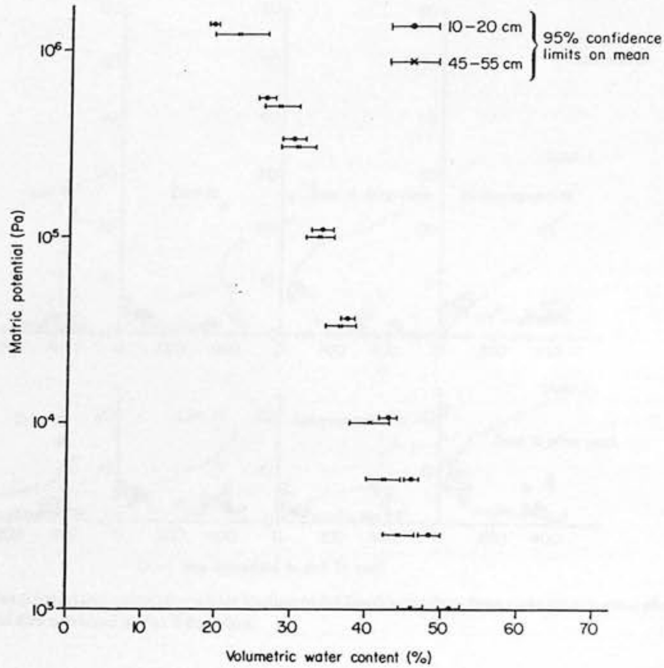

Fig. 5. Moisture release curves for A horizon and B horizon of Glencorse soil.

runoff was significantly contributing towards the rainfall recoveries, then this correction procedure underestimated nitrate losses, because the nitrate content of surface water was less. Unfortunately, it was not possible to identify the relative contribution of surface, ground and drain water movement into the plots.

Using these assumptions the nitrate loading from each plot was calculated for winter (Sept.–Feb.) and summer (Mar.–Aug.) periods. Results are presented in Figure 6. The following points emerge:

(1) there was no significant effect of fertilizer application

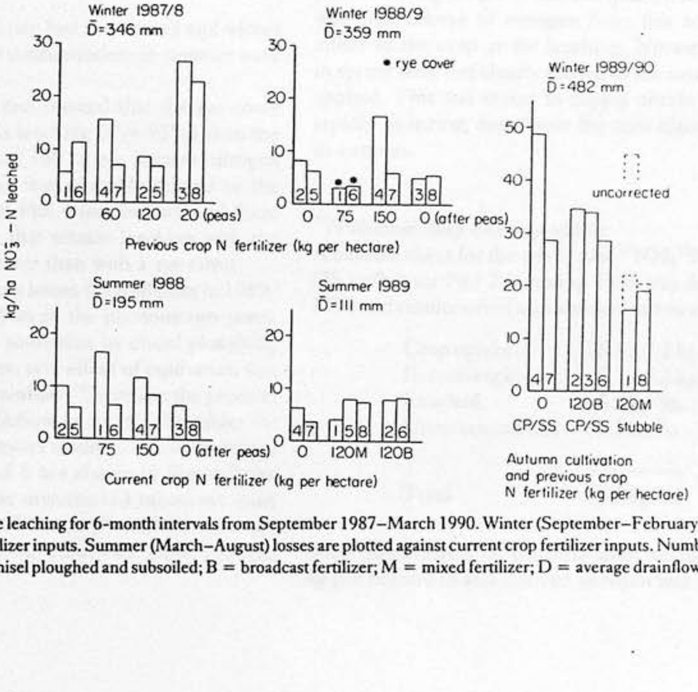

Fig. 6. Estimates of nitrate leaching for 6-month intervals from September 1987–March 1990. Winter (September–February) losses are plotted against previous crop fertilizer inputs. Summer (March–August) losses are plotted against current crop fertilizer inputs. Number in histogram is plot number. CP/SS = chisel ploughed and subsoiled; B = broadcast fertilizer; M = mixed fertilizer; D = average drainflow. All data corrected to Plot 7 drainflow.

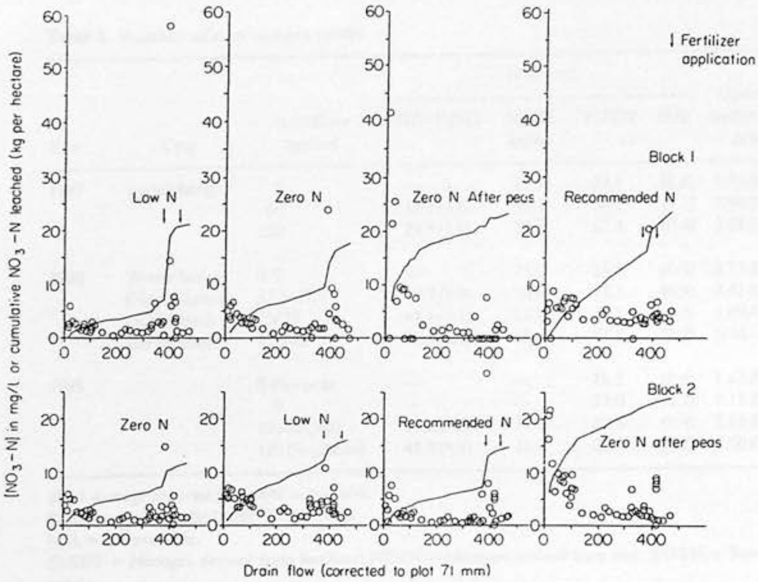

Fig. 7. Corrected nitrate concentration (circles) and cumulative nitrate loading (solid lines) in leachate from eight experimental plots, 22 September 1987–16 August 1988. All data corrected to Plot 7 drainflow.

amount on leaching losses either in the summer or in the winter following nitrogen fertilizer application;

(2) the incorporation of forage peas increased the leaching of nitrate by 10–15 kg N per hectare during the following winter, but caused virtually no increase thereafter. Using a studentized range test (Snedecor & Cochran, 1967, pp. 272–3), the leaching from the forage peas was significantly different from all other treatments ($P = 95\%$);

(3) the total amounts of nitrate lost in summer and winter were similar, but nitrate concentrations in summer were much greater;

(4) the studentized range test showed that the rye cover caused significantly less leaching ($P = 95\%$) than the plots in stubble following 150 kg per hectare nitrogen fertilizer. However, this was probably related to the anomalous behaviour of Plot 4 (see below), and there was no clear evidence that nitrate leaching with the land in stubble was greater than with a rye cover;

(5) the winter nitrate leaching losses from all plots in 1989/90 were much larger than in the previous two years. Losses were increased somewhat by chisel ploughing and subsoiling. However, one effect of cultivation was to delay the onset of drainflow. Therefore the procedure of using Plot 7 drainflow to correct the values for Plots 1 and 8 underestimates loading. The uncorrected loadings for Plots 1 and 8 are shown in Figure 6 for comparison. When these uncorrected values are used there is a significant difference ($P = 95\%$) only between the zero fertilized, cultivated plots and the stubble plots;

(6) the leaching losses from Plot 4 were consistently greater in all three winters than from all other plots receiving inorganic fertilizer, irrespective of the previous summer's fertilizer application.

The time course of nitrate leaching for the year 1987/88 on all eight plots was analysed using Plot 7 drainflow data (Fig. 7). There was a rapid initial loss of nitrate from the plots receiving the green manure (peas) treatment, but there was little release of nitrogen from this source in spring, either to the crop or for leaching. Nitrate concentrations in spring were not closely related to the amount of fertilizer applied. This soil seems to supply nitrate to drains more rapidly in spring, even from the zero nitrogen plots, than in autumn.

^{15}N Balance study and crop uptake

A balance sheet for the whole plot $^{15}NH_4{}^{15}NO_3$ application (75 kg/ha) on Plot 7 in spring 1988 was done on 18 May 1988 and results were (with standard errors where available):

Crop uptake:	25.8±3.2 kg/ha
Remaining in soil:	5.7±0.3 kg/ha
Leached:	6.0 kg/ha
Unaccounted for:	37.5 kg/ha
Total	75.0 kg/ha

Subsequent losses by leaching were negligible. Only 0.7 kg per hectare of soil-derived nitrogen was leached during

Table 5. Summary of crop nitrogen uptake

Year	Crop	N fertilizer applied	N uptake‡ NDFF (SE)	NDFS kg/ha	TOTN	(SE)	Grain dry matter yield (t/ha)	*n*
1987	Spring barley	0	—	22.1	22.1	(1.8)	1.04 (0.11)	4
		60	18.1 (2.6)	38.2	56.2	(4.3)	2.96 (0.30)	4
		120	29.9 (1.0)	32.5	62.4	(0.4)	3.08 (0.14)	4
1988	Winter barley	0/0	—	15.0	15.0	(0.9)	0.73 (0.07)	4
	(N applications	37.5/37.5	19.7 (1.0)	18.0	37.7	(0.4)	2.42 (0.18)	4
	on 29 March	75/75	44.1 (3.1)	32.0	76.1	(2.0)	4.09 (0.10)	4
	and 29 May)	75*/75	28.4 (0.4)	67.4	95.8	(2.2)	N/A—	2
1989		0 after peas	—	—	28.2	(0.4)	1.63 (0.03)	4
		0	—	—	23.0	(11.1)	1.11 (0.35)	2
		120 (mixed)	34.8 (9.6)	54.6	89.4	(9.9)	2.63 (0.27)	3
		120 (broadcast)	45.5 (4.5)	36.8	82.4	(9.2)	2.90 (0.13)	3

†bird damage reduced plot yield in one plot.
*only first split labelled with $^{15}NH_4{}^{15}NO_3$.
N/A = not available.
‡NDFF = Nitrogen derived from fertilizer; NDFS = Nitrogen derived from soil; TOTN = Total nitrogen uptake.

Table 6. Summary of nitrate leaching results from a selection of experiments in the UK

Measurement method	Cropping etc.	Nitrogen fertilizer (kg/ha); cultivation	Mean leaching losses (kg/ha)	No. years	Range (kg/ha)	Soil type	Reference
Catchment (Wytham, Oxford, England)	Mixed arable and grass	140	25	2	18–32	Mainly clay	White *et al.* (1983)
Catchment (Don, Aberdeen, Scotland)	Mixed arable and moorland	85	15	7	—	tills of variable composition	Edwards *et al.* (1990)
Lysimeter (Letcombe, Oxon, England)	Mixed arable	132	74.8	6	34–129	sandy loam	Webster *et al.* (1986)
		52	41	5	15–73	clay	
Lysimeter (Letcombe, Oxon, England)	Spring barley	0	83	4	59–125	chalk	Dowdell *et al.* (1984)
		80	74	4	55–113		
		120	84	4	65–121		
Lysimeters (Harpenden, Herts, England)	Bare fallow (1887/8–1893/4)	0	45	6	—	clay with flints	reported in Addiscott (1988)
	Bare fallow (1905/6–1922/12)	0	30.2	6	—		
Lysimeter (Craibstone, Aberdeenshire, Scotland)	Grass/ Arable rotation	0	7	6	3–20	'heavy loam'	Hendrick (1930)
		29.2 (FYM) + 14.1 (Ammonium sulphate)	9	6	3–24		
Drained plots (Brimstone, Oxfordshire) England)	Winter Cereals	175; P*	36	4	6–75	clay	Dowdell *et al.* (1987)
		175; DD*	32	4	3–66		
Drained plots (Jealott's Hill, Berkshire)	Cut grassland	250	4	3	0.5–6	clay	Barraclough *et al.* (1983)
		500	27	3	8–54		
		900	151	3	144–156		
Drained plots (Bush, Midlothian Scotland)	Winter and Spring barley	0	15	2	13–17	clay loam	This paper
		60–75	22	1	—		
		120–150	18	2	17–19		
	peas	After Peas	30	1	—		

*P = ploughed; DD = direct drilled.

the same period. Half the ^{15}N applied to Plot 7 disappeared, suggesting a large denitrification loss and/or uptake by the soil biomass. By harvest, the ^{15}N present in the crop in Plot 7 increased to 28.4 kg per hectare (Table 5). There was almost no ^{15}N in the leachate the following winter. This suggests that uptake by soil biomass followed by short-term release was not of major importance, although this cannot be ruled out.

Table 5 gives the details of nitrogen uptake from soil- and fertilizer-derived sources and crop yields for the three seasons covered. Recoveries of fertilizer nitrogen were poor (26–38%). After accounting for leaching, there is still a large amount of nitrogen unaccounted for. This strongly suggests that large losses by denitrification occurred. Yields, even with the recommended fertilizer amount, were small, but typical of much of the poorer quality arable land in Scotland. A clear priming effect occurred in all three seasons, as also observed by Smith *et al.* (1984). The largest priming effect occurred in 1989 when root development was severely impeded by drought.

Comparison with other work

Corrected estimates of mean annual nitrate leaching losses for the two completed years, 1987/88 and 1988/89, are compared in Table 6 with data from other experimental work on nitrate leaching. The losses were similar to those estimated in the catchment study in Aberdeenshire (Edwards *et al.*, 1990), but were generally less than those from experiments in southern England. Concentrations of nitrate were also much less because of the larger annual discharge/rainfall ratio (46% for Plot 7 in our experiment compared with 34% in the catchment experiment of White *et al.*, 1983).

The small contribution that residual soil nitrogen makes to leaching in the autumn also contrasts strongly with data from southern Britain. For example, Dowdell *et al.* (1987) reported nitrate concentrations in drainage water much greater than 10 mg per litre for the whole of the first 100 mm of winter drainflow on the clay loam soil (Denchworth series) at Brimstone. In our work, only in the plots where pea residues were incorporated did the drainflow concentration ever exceed 10 mg per litre in the autumn or winter. This is despite the fact that 27 mg mineral nitrogen per kilogram soil were measured on the zero nitrogen plots in September 1988. Overall losses were also considerably less than at Brimstone Farm, but were larger than the average annual losses (9 kg per hectare) measured in the leachate from intact soil lysimeters during a six year arable rotation (1921–1926) at Craibstone, Aberdeenshire (Hendrick, 1930). Annual drainflow was similar to that at Craibstone.

A strong influence of previous year's rainfall on nitrate leaching losses has been observed by Smith & Stewart (1989) and Addiscott (1988). The much larger losses in the winter of 1989/90 following a very dry summer agree with these observations and the effect of chisel ploughing and subsoiling on losses agree with the work of Dowdell *et al.* (1987).

CONCLUSIONS

The contribution of legume nitrogen to subsequent crops was much smaller than expected and legume nitrogen caused a significant increase (10–15 kg per hectare) in leaching losses after incorporation. Neither winter cropping nor the use of a winter cover crop was effective in decreasing nitrate leaching. Winter losses under winter barley were similar to those under a fallow stubble, but summer losses were slightly greater under winter barley than spring barley, because of earlier fertilizer application. If the soil organic matter content is stable, the difference between crop uptake and fertilizer application can be interpreted as total nitrogen loss, and this difference was larger under winter barley. However, cultivation should be as late as possible if a spring crop is to be grown to avoid mineralization while the soil is still warm. This makes it difficult to cultivate the soil in a suitable condition.

Decreasing fertilizer applications below the recommended level did little to decrease nitrate leaching. The strong priming effect of fertilizer additions meant that soil-derived nitrogen was used less efficiently if small fertilizer inputs were used. There was strong circumstantial evidence for large denitrification losses from this soil, although direct measurements were difficult to make because the 'acetylene-blocking' method does not work well on this soil.

The use of hydrologically isolated plots to measure nitrate leaching can be a useful technique on impermeable soils, but on this site variability in recovery of drainage water complicated quantitative measurement of nitrate leaching. We consider that the results obtained were nonetheless more reliable than those obtained by lysimetry because less disturbance was involved. Other methods (suction cups, soil sampling, etc.) where water flux is estimated indirectly, are also problematic, especially on heavy soils.

ACKNOWLEDGEMENTS

The authors acknowledge the technical support and advice of Mrs F. Wright, and the contribution towards the end of the experimental period of Mr B. Vivian. We also thank Mr F. Dry of the Macaulay Land Use Research Institute for the soil profile description.

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The Rothamsted soil and crop nitrogen service on Viewdata 1985–1989

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Abstract. During the four consecutive winters between 1984 and 1989 a computer simulation model was used to estimate the amounts of nitrogen in a cereal crop and available from soil to the crop after winter. The model does this by taking account of daily weather and by making simple assumptions about the starting conditions each autumn after the harvest of the previous crop. Some of the information which was given to farmers on viewdata systems is displayed, together with maps showing the average amounts of nitrogen in soil and crop in spring over 10 years in eastern England. This 10-year average is used as a baseline against which to judge the simulations in each of the four winters of our viewdata service.

INTRODUCTION

NITROGEN fertilizer applied to crops is currently a subject of some controversy in British agriculture. Farmers do not want to supply surplus nitrogen to their crops, yet more seriously they cannot risk the losses in yield and profit by not applying enough. Some of the nitrogen from fertilizer applications may leach to ground or river water (Foster *et al.*, 1982) and damage the natural balance in the environment. Leached nitrate finds its way into public drinking water, imposing extra costs on water companies which are charged by EC law (Council of the European Communities, 1980) to supply drinking-water that contains less than 11.3 mg nitrate-N per litre. It makes sense then, for farmers to ensure that their crops exploit soil reserves of mineral nitrogen to the full, and any system that can help them to estimate their fertilizer applications more accurately, making allowance for soil mineral nitrogen, is of value to the whole community.

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An analysis of the leaching of chloride tracer applied to pipe-drained plots using a coupled unsaturated–saturated zone model of solute transport

REDMAN, M.H.
PR.D.1992

A.J.A. Vinten, B. Vivian & M.H. Redman

Abstract. The modelling of leaching of nitrate and other solutes in artificially drained soils is complicated by the need to consider both unsaturated and saturated components of the system. This work is an attempt to couple together an unsaturated zone transport model (Addiscott & Whitmore, 1991) with a steady state travel-time model (Ernst, 1973) for the saturated zone. The model was tested using chloride leaching data from eight hydrologically isolated plots on a pipe-drained clay loam soil. Approaches to parameter estimation are discussed. Results were variable, for on some plots saturated zone travel time could be virtually neglected, whereas on other plots this component was apparently important.

INTRODUCTION

In recent years much research effort has been put into the problems of measuring and modelling water and solute transport through structured soils and other porous media (Addiscott *et al.*, 1986; Dyson & White, 1987; Steenhuis & Parlange, 1988; Barraclough, 1989a, 1989b). This is important for understanding and predicting the contamination of ground and surface waters by nitrate, pesticides, salts in irrigation water and accidental spills of toxic wastes, etc.

In many studies the one-dimensional, unsaturated zone transport is only a part of the system to be modelled. For example, heavy soils, which tend to be the most difficult to model, are also likely to be artificially drained. If so, a description is needed for transport to the ditch or pipe drainage system (e.g. Jury, 1975a). In this paper we couple together two mechanistic descriptions of solute transport to obtain an approximate description of solute transport in a pipe drained system on a clay loam arable soil. The two components of the system are:

(1) vertical transport through structured unsaturated soil under transient conditions;

(2) steady state advective transport in saturated homogeneous soil to gravel backfilled pipe drains.

Component (1) has been studied before on our experimental site (Vinten & Redman, 1990) and the model of Addiscott & Whitmore (1991) was used to predict vertical transport of bromide ions. An approach to modelling component (2) was proposed and tested by Jury (1975b), and Van Ommen (1985) proposed a model for a slightly different saturated zone geometry, which is used here.

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Coupled systems have been rigorously treated for water movement (e.g. Belmans *et al.*, 1983). Introducing solute transport rigorously is very difficult and some assumptions and approximations are necessary. Jansson & Andersson (1988) assumed piston displacement of the solute in the unsaturated zone and a uniform solute concentration at any time in any saturated horizontal soil layer. However, piston flow is unlikely to occur in the unsaturated zone of structured soils such as ours, and the solute travel time to drains from the water table will vary according to the distance from the nearest drain.

The coupled model used in this paper has been tested using leaching data from a chloride tracer experiment on drained field plots. Such a system for studying leaching of solutes has certain advantages. In many experimental systems, spatial variability of water flux and solute concentration means that large numbers of samples are needed to estimate field scale leaching (Richard & Steenhuis, 1988). The drainage system integrates all the spatial variability in the system giving, for a particular plot, a single determinate 'breakthrough curve'. If the travel time in the saturated part of the system is small or well defined, such a system can be used to calibrate and test models for vertical unsaturated flow and transport without recourse to exhaustive soil sampling (e.g. Barraclough, 1989a, 1989b). However, if travel times through the saturated part of the system are large, the delay between an experimental treatment on a plot and the response must be considered when interpreting leaching data. For example, Baker & Johnson (1981) applied fertilizer to experimental plots in alternate years only, and found that largest losses of nitrate-N from their system occurred in the years following fertilizer application. In interpreting the results of previous work on nitrate leaching (Vinten *et al.*, 1991), the extent of any such delays needs to be known.

THEORETICAL CONSIDERATIONS

Leaching of solute from soil surface to a water table at 30 cm is predicted by the layer model of Addiscott & Whitmore (1991). The details of this model are not reported here but other work (e.g. Vinten & Redman, 1990) has shown that the most critical parameter at this site is α, the proportion of mobile water present in a given soil layer which moves on to the next layer in a single day. This parameter strongly influences the extent to which 'bypass transport' of water and solute from the surface soil layer (0–5 cm) occurs during rainfall events. The model predicts that rapid bypass flow of excess water from the surface layer occurs if the capacity of the layer for mobile water is exceeded. If α is small (< 0.1, say), then many rainfall events will promote bypass flow. If α is large, then most rainfall will cause simple piston displacement of mobile water through the profile. Previous experimental work at this site has shown that this model predicts cumulative tracer leaching quite well when tracer is applied to microplots situated directly over drains (Vinten & Redman, 1990). Both 'best-fit' and independently estimated values of α are available from this work.

The daily output from the Addiscott model (drainflow and solute concentration) has been coupled to a model for steady-state convective displacement of solute in the saturated zone (Ernst (1973) as referred to by Van Ommen (1985) and Van Ommen *et al.* (1988)). In this model, the average concentration in ditches, following a unit step input of solute over the whole water table surface, is given by:

$$C_t^{out} = 1 - exp(-Rt/\varepsilon H) \tag{1}$$

where C_t^{out} = dimensionless mean concentration of solute in tile drains for a unit step input of solute
R = steady rainfall/drainage rate (mm/day)
ε = saturated zone porosity (dimensionless)
H = depth of saturated zone (mm)
t = time (days)

To couple this model to the unsaturated zone model it is necessary to discretize the expression in terms of daily drainage output from the unsaturated zone. For a step input solute concentration beginning on Day 1, the concentration of water entering the drains after N days, C_N^{out}, is given by:

$$C_N^{out} = C^{inp}\left[1 - exp\left(-\sum_{i=1}^{N} D_i/\varepsilon H\right)\right] \tag{2}$$

where D_i, i = 1 . . . N, are the daily drainage outputs from the Addiscott model
C^{inp} = input concentration to the saturated zone.

Consider a pulse of solute, C_I^{inp}, applied on day 1. The principle of superposition, used for calculating break-through curves for pulsed inputs, can be applied to determine the output response. The concentration following this one day pulse input is given by:

$$\mu(I, N) = \mu\left(D_I, \sum D_i\right)$$
$$= C_I^{inp}\left[exp\left(-\sum_{i=2}^{N}\left(D_i/\varepsilon H\right)\right) - exp\left(-\sum_{i=1}^{N}\left(D_i/\varepsilon H\right)\right)\right] \tag{3}$$

The contribution from each day's drainage from the unsaturated zone, between Days 1 and N, can be calculated in the same way, and the overall concentration in the drainflow is given by summation for $k = 1, \ldots N-1$:

$$C_N^{out} = \sum_{j=k+1}^{N} \mu(k, N)$$
$$= \sum_{j=k+1}^{N} C_j^{inp}\left[exp\left(-\sum_{i=j}^{N}\left(D_i/\varepsilon H\right)\right) - exp\left(-\sum_{i=j-1}^{N}\left(D_i/\varepsilon H\right)\right)\right] \tag{4}$$

The model of Ernst (1973) for steady-state ditch drainage was chosen in preference to the more accurate model (for our system) of Kirkham (1958), for steady-state drainage to pipes, as used by Jury (1975b), because the latter requires considerably more computation time to determine precise streamlines. This would have become prohibitive when doing tests to optimize the parameters, and a comparison showed that the two models gave similar results. The Ernst model has the advantage that radial flow is neglected, so the drain spacing, L, does not appear explicitly as long as $H/L < 0.1$ (Van Ommen *et al.*, 1988), which is usually true in our work. As the drainage layout in our system is quite complicated (Vinten *et al.*, 1991), this again makes calculations much simpler.

Figure 1 shows predictions given by the coupled model for selected values of εH and α. A fixed unsaturated zone depth of 30 cm has been used. When $\varepsilon H = 0.1$ mm, the results are effectively the same as for the unsaturated zone model alone. As εH increases, the output from the unsaturated zone is both smoothed and delayed as it passes through the saturated zone. When α is zero (matrix permeability negligible) surface applied tracer is rapidly leached to the saturated zone by bypass flow. As α increases, solute moves with the water through the unsaturated soil matrix and so takes longer to reach the saturated zone and the pipe drainage.

For structured soils, the physical meaning of the saturated zone storage term, εH, needs some discussion. If the macroporosity storage is small (1–2 mm), εH will be totally displaced during a storm event and most of the drainflow will be of 'new water' introduced in that storm. There will be little diffusive exchange of solute between the macropores and micropores in the saturated zone and εH will be essentially a macropore storage term. If the residence time of water in the saturated zone is much larger than the duration of single storm events, diffusive exchange will occur between the macropores and the micropores. This exchange

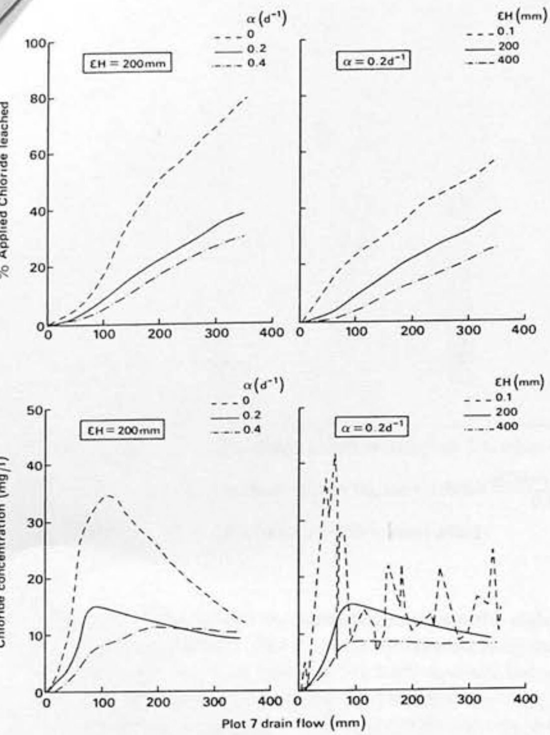

Fig. 1. Model predictions of leaching to pipe drains of a 100 kg/ha application of chloride tracer, using 1988/89 winter weather data:
(a) Cumulative leaching for εH = 200 mm and α = 0.0, 0.2 and 0.4 d^{-1};
(b) Cumulative leaching for α = 0.2 d^{-1} and εH = 0.1, 200 and 400 mm;
(c) Chloride concentration in drainage water for εH = 200 mm and α = 0.0, 0.2 and 0.4 d^{-1};
(d) Chloride concentration in drainage water for α = 0.2 d^{-1} and εH = 0.1, 200 and 400 mm.

may be almost complete if several days separate successive storm events, in which case the term εH represents total water storage in the saturated zone. Therefore size of the structural units in the saturated zone will influence the degree of equilibration, and hence the effective value of εH, as well as the macropore storage. Such a coupled model, therefore, is only a very rough approximation of processes occurring in a soil with a perched, transient water table. The model should be seen as a simplified 'grey box' model allowing some features of the system to be described mechanistically.

MATERIALS AND METHODS

Plot installation and hydrological measurements

The design and installation of the eight 300 m^2 hydrologically isolated plots used in this experiment are described in detail elsewhere (Vinten *et al.*, 1991). The drainage flow from each plot was measured using tipping bucket flow meters fitted with reed switches leading to integral counters. Water samples were accumulated from small subsamples (*c.* 10 ml) collected every second tip of the bucket; the collecting vessel was sampled and emptied about once a week.

Water content profiles in the top 40 cm of soil were measured on two plots on six dates using a γ-densitometry probe. Readings were made at 1, 3, 5, 7, 9, 12.5, 17.5, 22.5, 27.5, 32.5 and 37.5 cm depths. The collimator gave a resolution of 2 cm soil in the top 10 cm, and 5 cm below 10 cm depth. One of the plots was covered after a major storm event, so that internal drainage of the soil could be studied over a period of 70 days.

Potential evapotranspiration was obtained from regional data of the Ministry of Agriculture, Fisheries and Food.

Chloride tracer experiment

Potassium chloride fertilizer giving 135 kg chloride per hectare was applied to each of the eight plots on 12 September 1988 using two application methods:

(1) band placement at a nominal depth of 5 cm into the uncultivated soil using a 3 m direct drill; this was done on plots 2, 3, 5 and 7
(2) broadcasting, using the same drill, but with the pipes taking fertilizer to the drills removed so that metered fertilizer was spread fairly evenly on the soil surface; this was done on plots 1, 4, 6 and 8.

Soils were not sampled for residual chloride content before application. It was wrongly assumed that the effect of background chloride on results would be negligible. The chloride in soil extracts would usually have been at or below the detection limit of the chloride electrode. However, chloride concentrations were measured in drainage water the previous winter and also for several weeks before fertilizer application. From this information a 'background' chloride concentration attributed to residual chloride in the profile was calculated. We estimated that this declined linearly from 16.9 to 8.3 mg per litre over the experimental period.

At the end of the experiment (29 March 1989) two soil samples were taken at each of two depth increments (0–30 cm, 30–40 cm) from each plot, and extracted with water (1 : 2 soil : water ratio). The chloride content of these and all drainage water samples was measured using an Orion chloride electrode with a 90–01 single junction reference electrode. Unfortunately some of these determinations were below the linear range of the calibration (10 mg chloride per litre), so linear extrapolation of the calibration curve may have over-estimated the true concentration in these samples.

RESULTS AND DISCUSSION

Chloride and water balances

Table 1 gives details of chloride and water balance for the period 6 September 1988 to 29 March 1989. As set out by

Table 1. Water and chloride mass balance for the eight plots (6 September 1988–29 March 1989); rainfall = 476 mm, chloride input = 22.1 kg/ha

			Chloride leached		Chloride remaining	†Chloride recovery (%)	
Plot	Application method‡	Drainflow (mm)	Uncorrected (kg/ha)	Corrected (kg/ha)	in soil (kg/ha)	Uncorrected (%)	Corrected (%)
1	B	403	62.5	58.8	37.5	64	61
2	DD	251	58.2	80.9	22.0	51	66
3	DD	—	—	—	32.2	—	—
4	B	346	85.2	91.0	47.6	85	103
5	DD	746	183.4	87.9	93.6	177	115
6	B	305	77.4	93.7	95.8	110	121
7	DD	353	109.5	109.5	21.3	110	121
8	B	520	98.1	65.5	37.5	86	66
Mean		487	96.3	97.9	41.5*	94	88
S.E.		197	39.3	16.0	24.1–71.8	39	23
N		7	7	7	8	7	7

*Geometric mean of all eight plots. S.E. values are ±S.E. of geometric mean.

$$\dagger\text{Chloride recovery (kg/ha)} = 100\% \times \frac{\text{(Chloride leached + chloride remaining in soil)}}{\text{(Chloride in rain + chloride applied)}}.$$

‡B = Broadcast; DD = direct drilled.

Vinten *et al.* (1991) the water balance from the eight plots was quite variable. Plot 3 gave a very low recovery because of a pipe blockage and plot 5 a high recovery because of leakage from an isolation ditch and multiple counting on the electronic counter. Two estimates of chloride loss in drainage are shown – the uncorrected values and values calculated assuming Plot 7 drainflow applies. Plot 7 gave recoveries of rainfall which were very consistent, and this plot was therefore used as a 'benchmark'. The chloride recovery in the soil at the end of the experiment has a large uncertainty because many of the 2:1 extracts of soil gave chloride concentrations less than 10 mg per litre, so some extrapolation of the calibration curve was necessary. Assuming the soil value is reasonable, and allowing for rainfall inputs, the average measured recovery of chloride was over 88%. However, two unmeasured components are not included in this mass balance. First, initial soil chloride content was not measured. The background chloride estimated from drainage water chloride concentration the previous winter (see Materials and Methods) amounted to an additional 44 kg per hectare. Second, there was some deep percolation in these plots, shown by the incomplete recovery of incident winter rainfall; even after allowing for evaporation, about 12% of incident rain was unaccounted for in Plot 7. If deep percolating water had similar chloride concentration to the water entering the pipe drains, the leaching of chloride increases by about 18 kg per hectare. The effect of these two corrections is to decrease the mean recovery to 75%.

Figure 2 gives the mean cumulative loss of chloride by leaching for the six plots which gave reasonable recovery. Losses on direct drilled and broadcast plots were similar, so all data have been pooled, using drainflow from Plot 7 to calculate the loading.

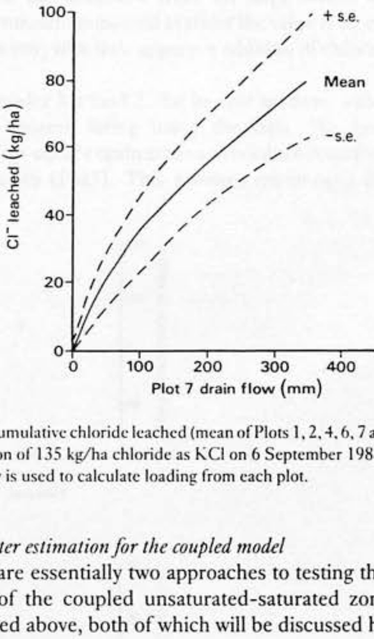

Fig. 2. Cumulative chloride leached (mean of Plots 1, 2, 4, 6, 7 and 8) after application of 135 kg/ha chloride as KCl on 6 September 1988. Plot 7 drainflow is used to calculate loading from each plot.

Parameter estimation for the coupled model

There are essentially two approaches to testing the applicability of the coupled unsaturated-saturated zone model described above, both of which will be discussed here.

(1) measure the two main model parameters: α, the soil permeability, and εH, the saturated zone storage depth. We used values for mobile, retained and ion exclusion fractions of 0.127, 0.287 and 0.087, respectively. We assumed a constant value for β, the parameter describing solute diffusion into immobile water, of 0.2. For this soil, the predictions were found to be rather insensitive to variation in these parameters (Vinten & Redman, 1990);

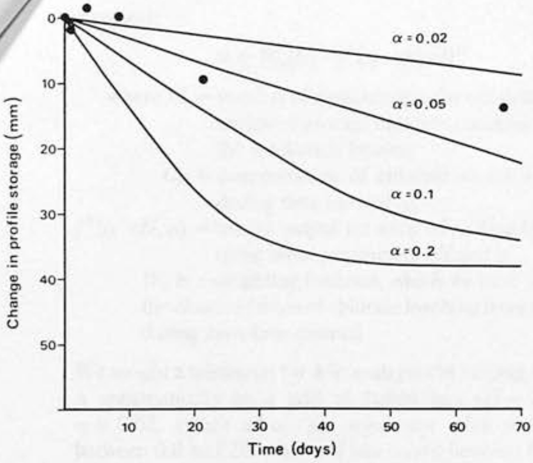

Fig. 3. Changes in soil water storage in a covered plot during the period 29 January 1990 to 6 April 1990, compared with modelled changes using SLIM. ● Measured values; —— Modelled values. Figures on curves represent values of the parameter α in the model.

(2) obtain the parameters by a least-squares fitting routine from the data on chloride leaching. This type of approach is known as the inverse method (Yeh, 1986).

We attempted to estimate α by measuring the internal drainage of the soil by covering a fully wetted soil profile and measuring the change in its water content with time using a gamma probe. Figure 3 compares the modelled and measured changes in soil moisture storage between 0 and 30 cm. Measuring such small changes in storage was difficult, but the results suggested a value for α of 0.02–0.05. This compares reasonably with estimates of 0.08–0.14 made by least squares fitting of daily drain outflow to the Addiscott model (Vinten & Redman, 1990).

The saturated zone storage depth, εH, was estimated by calculating the amount of storage in the soil during storm events from hydrograph data. During the largest storms for which data were available, the storage was only 3.3 mm. This gives an upper estimate of drainable porosity, as it also includes depression storage and the water required to replace pre-storm deficits in the unsaturated zone. This figure includes only the macroporosity and if the storage term is only a few millimetres, the velocity of water in the transporting macropores will be too large for much diffusive exchange to occur during storm events. In winter months, when the model was tested, convective movement of water and solutes from macropores into the bulk soil should also be minimal. Chemograph data (Fig. 4) support this interpretation of the saturated zone storage term. A small storm (Fig. 4a) caused no change in the chloride concentration of the drainage water, but a large storm (Fig. 4b) caused dilution. During most of the drainflow from the large storm, the chloride concentration remained at about the value to be expected in rainwater, with little apparent addition of chloride from the soil.

We now consider Method 2, the inverse method, which involves least-squares fitting using the data. We have followed the least-square optimization procedure described by Jury & Sposito (1985). This involves minimizing the

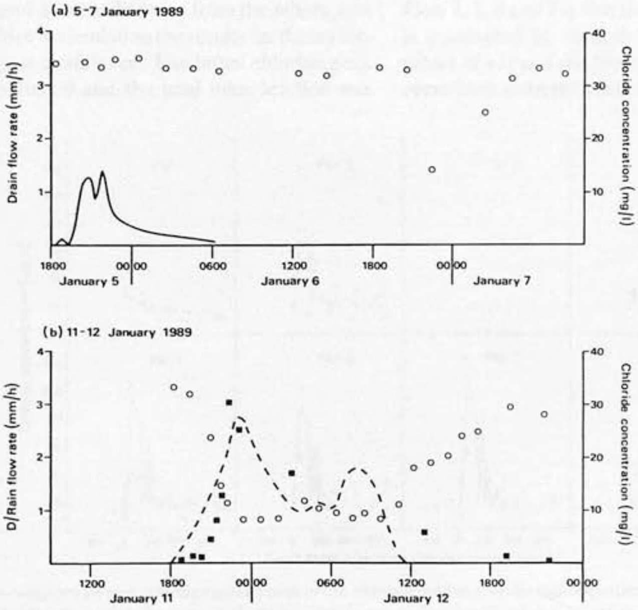

Fig. 4. Chemographs for two storm events in winter 1988/89: (a) 5–7 January 1989. ○ – $[Cl^-]$ in drainage water; solid line – drainflow rate; (b) 11–12 January. ○ – $[Cl^-]$ in drainage water; ■ – drainflow (incomplete); —— rainfall for nearby site.

function:

$$\phi = W_K[C_K - f^K(t_K : \varepsilon H, \alpha)]^2 \qquad (5)$$

where M = number of time intervals for which the flow-weighted average chloride concentration in the leachate is known;

C_K = concentration of chloride in the leachate during time interval t_K;

$f^K(t_K : \varepsilon H, \alpha)$ = model output for each of t_K time intervals using input parameters εH and α;

W_K is a weighting function, which we have taken as the observed mass of chloride leaching from the plot during each time interval.

We sought a minimum for ϕ in each plot by varying εH and α systematically on a grid of dimensions $\varepsilon H = 20$ and $\alpha = 0.02$, except where the minimum value of εH lay between 0.0 and 20, when εH was varied between 0.1 and 20.1 in intervals of 10.

Results of this least-squares method of parameter identification are given in Table 2. Figure 5 shows measured chloride concentrations on the eight plots and the simulated values based on the best-fit parameter estimates. The measured chloride concentrations in drainflow for Plot 7 prior to chloride application are also shown. A correction for this background chloride concentration was made to the model predictions (see Materials and Methods).

The parameter values obtained by the least squares method for Plots 2, 5, 6 and 7 were reasonably consistent, could be estimated with a good degree of confidence, gave reasonable simulations and were similar to those estimated independently. However the breakthrough curves for three of the plots (1, 4 and 8) were different from the others, and this led to difficulties in simulating the results for these plots with any accuracy or confidence. The initial chloride peak was far less pronounced and the total mass leached was much less on Plots 1 and 8. For these data, the response surface of ϕ to variation in εH and α was quite flat. Figure 6 shows the 95% confidence limits on joint estimates of (εH, α) for three plots. The indeterminacy of the optimum combination of parameter values for Plot 8 and, to a lesser extent Plot 4, is clear. However, for Plot 7 the optimum values are well defined.

The physical interpretation of the results of this work for Plots 2, 5, 6 and 7 is that the travel time of chloride to drains is dominated by vertical transport processes. The small values of εH and the 'noisy' character of the breakthrough curve both indicate a minimal delay in the saturated zone.

Table 2. Estimation of parameters α and εH in solute transport model

Plot†,‡,§	Cl^- application method*	$\alpha(d^{-1})$	εH (mm)	Predicted total Cl^- leached (kg/ha)
1	B	0.18	> 1000	60.9
2	DD	0.18	0.1	117.8
3	DD	—	—	—
4	B	0.26	80	104.0
5	DD	0.12	0.1	115.5
6	B	0.14	0.1	116.4
7	DD	0.10	0.1	115.6
8	B	0.14	640	71.3
Independent estimation (Plot 7)		0.08–0.14¶	3	117.8–130.0

Parameter estimates found by grid search to nearest 20 mm (for εH) or 0.02 d^{-1} (for α). 0.1 is the minimum value of εH tried.

*B = broadcast; DD = drilled with a 3 m Moore direct drill to nominal depth of 5 cm.

†Plots 2, 3, 4, 5, 7 and 8 were in stubble till mid-March 1989.

‡Plots 1 and 6 were rotavated in August 1988 and a rye cover crop was grown.

§Predictions made on the basis of predicted drainflow using Plot 7 overall recovery to estimate deep percolation.

¶Independent estimates of α from Vinten *et al.* (1990).

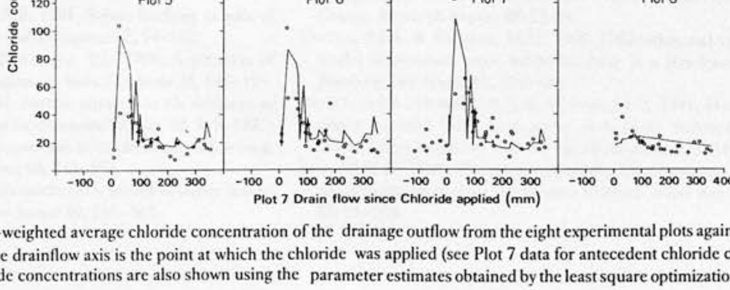

Fig. 5. Measured flow-weighted average chloride concentration of the drainage outflow from the eight experimental plots against cumulative Plot 7 drainflow. Zero on the drainflow axis is the point at which the chloride was applied (see Plot 7 data for antecedent chloride concentrations). The model predicted chloride concentrations are also shown using the parameter estimates obtained by the least square optimization method (see Table 2).

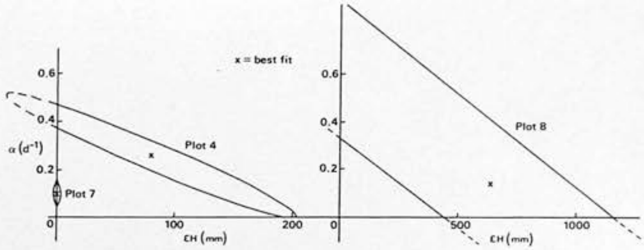

Fig. 6. Confidence ellipses (95%) for the estimates of α and εH using the least squares optimization method. The optimum is marked as x.

However, for Plots 1 and 8, and to a lesser extent Plot 4, some process is causing considerable depression, smoothing and/or delay of the breakthrough curve. By the inverse method we cannot clearly distinguish the reasons for this different behaviour, and more work is needed to establish whether it is saturated zone or unsaturated zone transport that is different on these plots.

CONCLUSIONS

Independent and least squares methods of parameter estimation gave similar and physically reasonable parameter values for the proposed chloride leaching model in four of the experimental plots. This suggests that prediction of solute leaching would not be much less accurate on this soil if only the unsaturated zone model is used without allowance for saturated zone travel time. However, on three of the plots the least squares parameter estimation suggested a considerable delay of solute in the saturated zone. A more detailed sensitivity analysis involving more than two parameters may be needed. More work is also needed to describe processes causing solute dispersion in the saturated zone and on the effects of a fluctuating water table.

ACKNOWLEDGEMENTS

We thank Mrs L. Johnstone for preparing the manuscript and K. Henshaw and M. O'Sullivan of the Soil Engineering Department, Scottish Centre of Agricultural Engineering, for help in use of the γ-densitometry probe.

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